PB82-196874

The Monitoring of the Restorational Dredging of Collins Lake, Scotia, New York

Union Coll.
Schenectady, NY

Prepared for

Corvallis Environmental Research Lab., OR

Feb 81

U.S. Department of Commerce
National Technical Information Service
NTIS

PB82-196874

EPA-600/3-81-017
April 1982

THE MONITORING OF THE RESTORATIONAL DREDGING OF COLLINS LAKE, SCOTIA, NEW YORK

by

Carl J. George
Peter L. Tobiessen
Department of Biological Sciences

and

Phillip D. Snow
Thomas K. Jewell
Department of Civil Engineering
Union College
Schenectady, New York 12308

R804572

Project Officer

Frank Stay
Environmental Research Laboratory
Corvallis, Oregon 97333

ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CORVALLIS, OREGON 97333

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA 22161

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO.	2.	3. RECIPIENT'S ACCESSION NO.
EPA-600/3-81-017	ORD Report	PB82 196874

4. TITLE AND SUBTITLE	5. REPORT DATE
The Monitoring of the Restorational Dredging Of Collins Lake, Scotia, New York	February 1981
	6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S)	8. PERFORMING ORGANIZATION REPORT NO.
Carl J. George, Peter L. Tobiessen, Phillip D. Snow and Thomas K. Jewell	
9. PERFORMING ORGANIZATION NAME AND ADDRESS	10. PROGRAM ELEMENT NO.
Union College Schenectady, New York 12308	
	11. CONTRACT/GRANT NO.
	R804572
12. SPONSORING AGENCY NAME AND ADDRESS	13. TYPE OF REPORT AND PERIOD COVERED
Environmental Research Laboratory Office of Research and Development U.S. Environmental Protection Agency Corvallis, Oregon 97333	extramural - final 1976-1979
	14. SPONSORING AGENCY CODE EPA/600/02

15. SUPPLEMENTARY NOTES

Frank Stay, Project Officer. FTS 420-4879 (503-757-4879)

16. ABSTRACT

Collins Lake (24.3 ha) in the Village of Scotia, New York, was hydraulically dredged to a depth of 3 m during the warmer months of 1977 and 1978. About 52,000 m^3 of organic sediment were removed from 2.6 ha of lake bottom to an adjacent sedimentation basin with the supernatant water returning to the lake. The curly leaved pondweed, Potamogeton crispus, especially abundant in the lake, did not recolonize the dredged areas through the study period. Other biotic, chemical and physical features of the lake remained relatively unchanged.

Alteration of the lake level caused by changes in the outlet resulted in vegetational changes in the littoral zone, such as the decline of the tuckahoe, Peltandra virginica and the increase of the spiked loosestrife, Lythrum salicaria. In some places, dense stands of the loosestrife blocked access to the lake. Water and phosphorus budgets were also developed suggesting a net loss of phosphorus from the lake. The sediments were most likely the main source.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group

18. DISTRIBUTION STATEMENT	19. SECURITY CLASS *(This Report)*	21. NO. OF PAGES
Release to public	Unclassified	
	20. SECURITY CLASS *(This page)*	22. PRICE
	Unclassified	

EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE

DISCLAIMER

This report has been reviewed by the Corvallis Environmental Research Laboratory, U.S. Environmental Protection Agency, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the U.S. Environmental Protection Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

ABSTRACT

Collins Lake (24.3 ha) in the Village of Scotia, New York, was hydraulically dredged to a depth of 3 m during the warmer months of 1977 and 1978. About 52,000 m^3 of organic sediment were removed from 2.6 ha of lake bottom to an adjacent sedimentation basin with the supernatant water returning to the lake. The curly leaved pondweed, Potamogeton crispus, especially abundant in the lake, did not recolonize the dredged areas through the study period. Other biotic, chemical and physical features of the lake remained relatively unchanged.

Alteration of the lake level caused by changes in the outlet resulted in vegetational changes in the littoral zone, such as the decline of the tuckahoe, Peltandra virginica and the increase of the spiked loosestrife, Lythrum Salicaria. In some places, dense stands of the loosestrife blocked access to the lake.

Water and phosphorus budgets were also developed suggesting a net loss of phosphorus from the lake. The sediments were most likely the main source.

This report was submitted in fulfillment of grant R804572 by Union College under the sponsorship of the U.S. Environmental Protection Agency. This report covers a period from 1976 to 1979 and work was completed as of 1979.

CONTENTS

	Page
Introduction	1
Conclusions	3
Recommendations	3
Methods	4
Results	8
Hydraulic Dredging	8
Morphometry	11
Water Budget	11
Sediments	11
Water Chemistry	18
Vascular Plants	29
Phytoplankton	40
Zooplankton	44
Fishes	50
Discussion	57
Physical Changes	57
Chemical Changes	58
Biological Changes	59
Phosphorus Budget	60
Phosphorus and Plant Growth	66
References	70
Appendices	73

Preceding page blank

FIGURES

Number		Page
1	Collins Lake setting	2
2	Collins Lake sampling stations and transects	5
3	Sedimentation basin for Collins Lake dredged sediments	9
4	Bathymetry of Collins Lake before and after dredging	12
5	Hypsographic curves before and after dredging	14
6	Temperature isopleths at West Station	20
7	Percent saturation of dissolved oxygen isopleths at West Station	21
8	Percent saturation of dissolved oxygen at North and South Stations	22
9	Isopleths of pH at West Station	24
10	Ammonia at North and South Stations	25
11	Ammonia isopleths at West Stations	26
12	Nitrate at North and South Stations	27
13	Nitrate isopleths at West Station	28
14	Orthophosphate at North and South Stations	30
15	Orthophosphate isopleths at West Station	31
16	Total phosphorus at North and South Stations	32
17	Total phosphorus isopleths at West Station	33
18	Map of *P. crispus* extent, transects and dredged areas	35
19	Biomass of *P. crispus* versus time	36
20	Positions of vegetation zones along the SE transect	39
21	Chlorophyll *a* at West Station	41
22	Chlorophyll *a* at South Station	42
23	Secchi disc depth at West and South Stations	43
24	*Pediastrum simplex* densities at West Station	45
25	Percent major taxa of phytoplankton at West Station	46
26	Percent major taxa of phytoplankton at South Station	47
27	Percent rotifers, cladocerans and copepods at West Station	48
28	Percent rotifers, cladocerans and copepods at South Station	49
29	Zooplankton species diversity at West Station	51
30	Gonadosomal index for yellow perch	52
31	Mass balance diagram for phosphorus in Collins Lake	61
32	Total phosphorus at 0.5 m and 3.0 m West Station	68

TABLES

Number		Page
1	Removal of nutrients and turbidity by sedimentation basin	10
2	Morphology data - before and after dredging	13
3	Water budget for Collins Lake, Scotia, N.Y.	15
4	Sediment core analyses	16
5	Concentrations of metals in sediment samples	17
6	Maximum and minimum values for chemical parameters	19
7	Total Kjeldahl Nitrogen for selected stations	29
8	P. crispus biomass (gm ODW/m^2) estimates for 1979	37
9	Common vascular plants according to vegetation zones before and after dredging	40
10	Number and type of male and female yellow perch and golden shiner-catch per net day	50
11	Morphologic parameters for perch and shiners	53
12	Condition indices for yellow perch	54
13	Macrophage centers (number per mm^2) in liver and spleen of yellow perch	55
14	Ceroid indices for various organs in yellow perch and golden shiners	56
15	PO_4 budget for Collins Lake	63
16	Mass balance equation for phosphorus	64
17	Calculations for lake phosphorus using mass balance equation	65
18	Biomass and phosphorus content of P. crispus	67

ACKNOWLEDGEMENTS

Many undergraduate students have participated as tutorial researchers or technicians in this project and we are exceedingly proud of their contributions (see Appendix A). We have also received the guidance of many persons within and outside of the College to whom we are especially grateful. Wolfgang Fuhs--physical and chemical matters; Susan Allen, Edward Mills and Joseph Makariwicz--plankton sampling and analysis; Eugene Ogden--macrophytes; Willard Roth, George Smith, Paul Yevich and Edward Kuzia--histology; Helena Birecka, Christopher Jones and Robert Schaefer--chemical matters; Paul Mason--computer programming; Louise DiDonna--secretarial; Katherine Martin and Mary Werner--graphics. Calvin Welch and William Adams, who were in charge of the dredging phase of the project, were essential in providing access and technical support for various aspects of the study.

Finally, we are pleased to dedicate this work to Mr. and Mrs. Paul H. Dietershagen and the other citizens of Scotia who made their lake available for study.

INTRODUCTION

Eutrophication is a natural process, but it is often accelerated in urban lakes due to high nutrient loading or other "cultural" inputs. The result is a build up of nutrient rich sediments which act as a recycling nutrient source, adding to the productivity of the lake. Since urban lakes will most likely become even more heavily used in the future by the local population due to cost constraints on travel, there is interest in restoring them by slowing the rate of sedimentation and/or regaining lake volume.

The successful reclamation of an urban lake often requires a reduction in the nutrient pool stored in the sediments. Of the many techniques available to remove lake sediments hydraulic dredging may be the least disruptive of the lake ecosystems (Dunst et al. 1974; Theis and DePinto 1976). Lake Tummen in Sweden, where 400,000 m^3 of sediment were removed, is one of the more remarkable examples (Bjork 1977; Gelin 1978).

The detailed consequences of hydraulic dredging, apart from the simple increase of lake volume, however, have been considered in relatively few instances. The cited work on Lake Tummen, a study of Stone Lake in Michigan (Theis and DePinto 1976), and a number of studies sponsored by the Environmental Protection Agency and now under way (Peterson 1979) are examples. In general, much more work is needed especially in regard to the evaluation of: 1) nutrient mobilization and its partitioning by plankton and macrophyte populations; 2) the role of dredged areas in supporting macrophytes; and 3) the mobilization of potentially dangerous materials such as heavy metals and chlorinated hydrocarbons.

The objective of this study was to evalute the impact of hydraulic dredging on Collins Lake. The study ran from the summer of 1976 through August 1979, during which the lake was dredged in the summers of 1977 and 1978. For a discussion of the dredging per se, including the rationale for the use of dredging as a restorative strategy, the mechanics and engineering of the dredge and sedimentation basin the reader is referred to an earlier report (Welch et al. 1979).

The study site was Collins Lake (Figure 1), in the Village of Scotia, Schenectady County, New York at north latitude 42° 49' 36" and west longitude 73° 57' 24". It is thought to be of kettle hole origin and modified by flooding action of the Mohawk River. The primary inflow for the lake is a series of springs along the western and northern shores. A storm water sewer also contributes to the lake during wetter periods. The l[illegible] drains at the eastern end through a 90 cm diameter culvert and flapper valve into Collins Creek which flows into the Mohawk River. The average annual precipitation for the area is 91.39 cm with 137.2 cm of snowfall. Average annual sunshine is about 54% of that potentially available. The area of the lake is 24.4 ha.

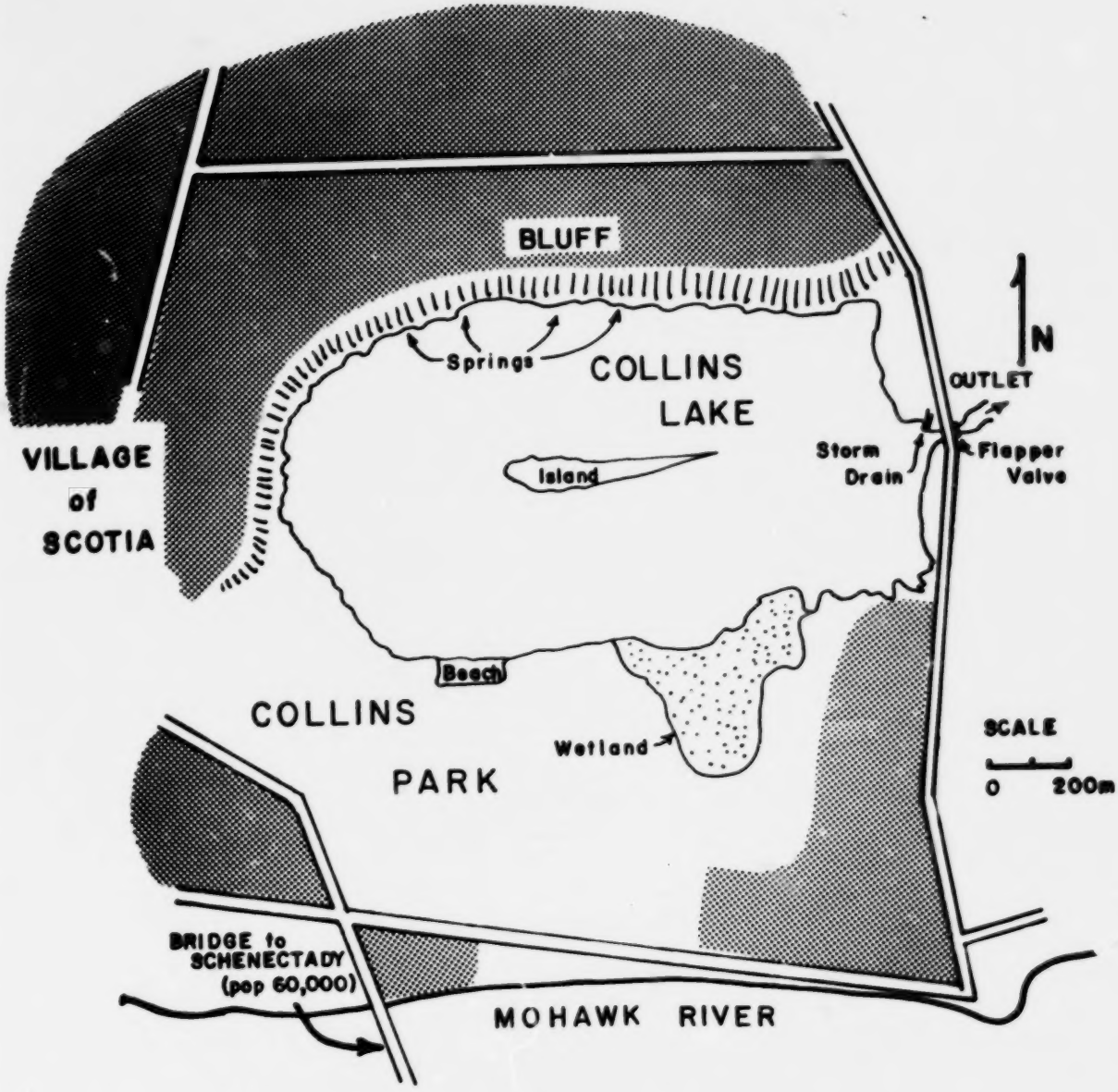


Figure 1. Collins lake setting. Stippling represents residential areas.

CONCLUSIONS

Hydraulic dredging has been shown to be a viable lake restoration technique for Collins Lake. Approximately 53,000 m^3 of nutrient rich, organic sediments were removed and the curly leaved pondweed, Potamogeton crispus, which previously had grown at problem densities over most of the lake, did not recolonize the dredged areas during the study period. Throughout the dredging period recreational use of the lake was not adversely affected. The resulting deepening of the lake and reduction in P. crispus growth have enhanced the swimming and boating possibilities for the lake.

The limited areal extent of dredging (2.6 ha or about 10% of the lake area) along with the brevity of the study period have not permitted satisfactory definition of possible long term shifts in ecosystem equilibrium resulting from hydraulic dredging as a restorative technique. However, the observed lack of significant changes in short term chemical and biological indicators suggests that careful application of dredging may not be overly disruptive to the non-benthic lake ecosystem. During our study we found no increase in phosphorous content of the water, no change in chlorophyll a content, no shift in the dominant forms of phytoplankton during the growing season (there was a shift during the winter months), no changes in dominant zooplankton or in zooplankton diversity, and only minor changes in condition indices of the two species of fish studied.

Some changes in the lake ecosystem were observed, however, most of which are attributable to the lowering of the lake by about 50 cm, i.e. not to dredging. There were notable zonal and density changes for various plant species of the littoral zone, such as the increased abundance and areal extent of Lythrum salicaria, and Najas minor and the decline of Peltandra virginica. These changes were of limited areal extent and may be reversed when the original lake level is re-established.

RECOMMENDATIONS

It has been difficult in the present study to separate the effects of dredging per se from those resulting from the lowering of the lake level and the seasonal and annual variations typical of natural lakes. Clearly this study is best considered as a beginning, for only continued evaluation would document the long term effectiveness of hydraulic dredging as a restoration tool.

Given the generally successful outcome of the dredging phase of the study, further dredging should be undertaken by the Village to improve overall lake qualities. The sediment now held in the sedimentation basin could be removed permitting receipt of additional lake sediment. Stumps obstructing the operation of the dredge should be winched out of the lake.

METHODS

Sampling of Collins Lake for most parameters took place at four lake stations and a ground water station (Figure 2) commencing in July 1976 and concluding in August of 1979. The four stations were chosen so that they would be representative of the overall lake environment. North and South stations were originally located in parts of the lake that were to be dredged. East station was also located in the shallow portion of the lake in an area that was not to be dredged, to act as a control for North and South stations. West station was located in the deep portion of the lake.

Samples were gathered from selected depths every two weeks, except where thin ice prevented access. Data collecting occurred on an appropriate ad hoc basis for water-sediment mix of the sedimentation basin, turbidity in the vicinity of the dredge, water quality of marginal springs, heavy metals and certain chlorinated hydrocarbons associated with the sediments, aquatic macrophytes and the higher plants of the shore zone.

East and South stations were moved slightly. When the lake was lowered water depth at East station decreased enough so that it was impossible to gather an undisturbed sample. Because of changes in dredging plans (Welch et al. 1979), it was necessary to move South station approximately 30 m to the west so it would be in the dredged area. This was accomplished in November of 1977.

Water chemistry samples were collected with a 2 liter Van Dorn bottle within a few hours of noon, transferred to standard bottles in the field, fixed if necessary, and returned to the laboratory for analysis. Dissolved oxygen and temperature measurements were done in situ and transparency (Secchi depth) measurements were done at the same time. Samples were kept in plastic bottles under refrigeration and analyzed usually within 24 hours. All chemical measurements were conducted according to Standard Methods (14th ed).

Vegetation surveys were taken in and around the lake before and after dredging. In addition, four transects were established for both qualitative and quantitative surveys (Figure 2). Quantitative sampling centered on the P. crispus, and was carried out using an open ended wire mesh cylinder to delineate a known sample area (0.75 m^2). Plants within this cylinder were collected using a long handled garden rake equipped with a cutting edge wired to the tines and a wire mesh scoop to collect the cut and dislodged plants. The dredged area was too deep to use the cylindrical sampler effectively, so two alternative sampling techniques were used: 1) the rake was dragged for an approximated distance along the bottom; 2) quadrats were sampled by divers. Tests to compare effectiveness of the wire cylinder and rake technique showed there was agreement within 10% between the diver quadrats and the cylinder

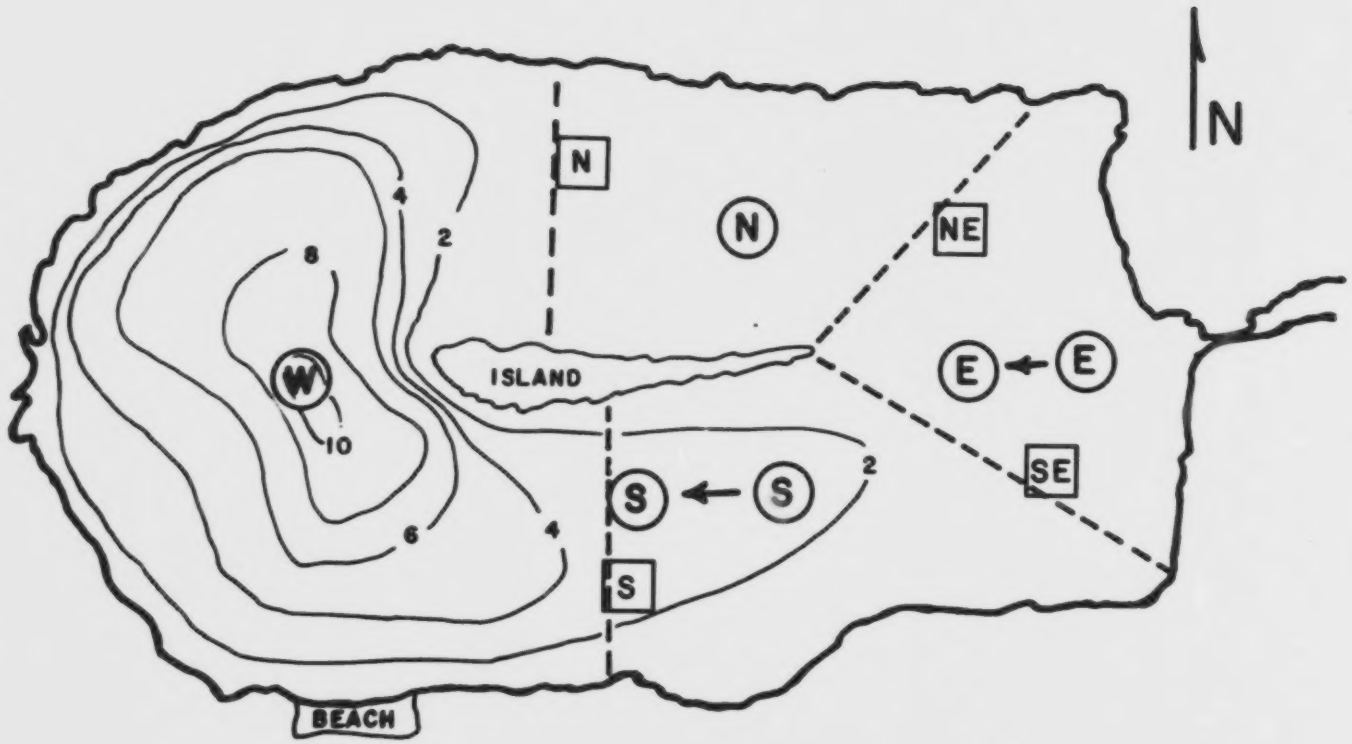


Figure 2. Collins Lake sampling stations (circles) and vegetation transects (boxes). Stations E and S were moved during the course of the study. Depth contours are in meters. A ground water-station sampled a spring to the north of the lake.

method. Sampled plants were then counted (number of apices), dried and weighed. Phosphorus concentrations of P. crispus were determined using an acid digestion and the molybdenum blue method (Allen 1974).

Plankton were collected with a Van Dorn bottle every other week from 0.5 m at East and North stations, 0.5 m and 1.5 m at South and 0.5 m, 1.5 m, 3.0 m, 6.0 m and 8.0 m at West Station. Phytoplankton were concentrated by settling from one liter of lake water fixed with buffered formalin (5% v/v) and counted on an inverted microscope.

Lake water was filtered and the particulate matter extracted with acetone within 12 hrs of collection for chlorophyll a analysis according to the methods in Vollenweider (1974). Initial chlorophyll a values (until 7/20/77) did not include a correction for phaeophytin, whereas later values did. In most cases, this correction was not greater than 5% of the uncorrected value.

Zooplankton were concentrated from 4.4 ℓ lake water using a 35 μ nylon mesh sieve. Samples were stored in 5% buffered formalin (v/v) and counted using an inverted microscope. During the summer and fall of 1978, the diurnal distribution of the zooplankton was studied. Samples were taken every 4 hrs beginning at noon on four dates including both thermally stratified and unstratified conditions. Depths of 0.5, 1.0, 1.5, 2.0, 2.5, 3.0, 3.5, 4.0, 6.0 and 8.0 m. Zooplankton diversity was determined from samples collected with a No. 10 plankton net submerged to 6 m and raised to the surface at West Station. For each date 400 individuals were tallied and a Shannon-Weaver index calculated.

Specimens of the yellow perch, Perca flavescens, and the golden shiner, Notemigonus crysoleucas, were collected every two weeks using gill nets comprised of 19, 25, 132, 138, and 144 mm square mesh panels 7.6 m long and 1.2 m deep. Nets were set in the afternoon, recovered the next morning, and catch per unit of effort was calculated. Specimens of other species were noted and released. During the warmer months, fish were returned to the laboratory in the gill netting which fostered their survival until processing. During the colder months, specimens were immediately removed from netting and placed in water to prevent freezing. At the laboratory specimens were measured for total length (L), the whole body and gonads weighed to the nearest gram (W_T,W_G) for the calculation of condition indices:

$$\frac{W_T}{L^3} \times 10{,}000 \quad \text{or} \quad \frac{W_T - W_G}{L^3} \times 10{,}000$$

Generally, the numerical value of the condition index is thought to increase with health and available food supply.

Scales were taken from the left side in pectoral area, below the lateral line of the yellow perch and above the lateral line of the golden shiners. Mobility of the heart was noted and the specimen was recorded 'alive' if motion was observed and 'dead' if not. Sex was determined and the spleen, liver, and gonads removed and fixed.

After fixation, the liver (WL) and gonads were weighed to 0.001 g to generate hepatosomal (HI) and gonadosomal (GI) indices:

$$HI = \frac{W_L}{W_T - W_a} \qquad GI = \frac{W_G}{W_T} \times 100$$

Both are considered useful indicators of physiological conditions (Larson 1973; Heidinger and Crawford 1977). Blocks of tissue from standard location for liver, spleen and kidney were then removed and processed histologically (Pearse 1968). The sections were then examined, giving particular attention to the macrophage centers and their ceroid content. These features were selected for special study because they have been considered indicitave of the traumatic history of the individual (Gyorgy and Goldblatt 1941; Lillie et al. 1941; Gyorgy 1944; Wood and Yasutake 1956; Hartroft and Porta 1968; Bongner and Ellis 1976; Ellis, Monroe and Roberts 1976; Sorenson 1976).

The number of centers by three size categories; i.e., small, less than 0.005 mm; medium, greater than 0.005 but less than 0.01 mm and large greater than 0.01 mm, was determined (at 80 power magnification) for 15 fields and averaged. The resulting values were then converted to macrophage center density and to total cross sectional area (and volume) using average diameters for each of the three size categories. Ceroid is known to increase with age (Bjorkerud 1963; Reichel 1968; Reichel et al. 1968; Munnell and Getty 1968; Donato and Sohal 1978), so the percentage ceroid volume was divided by the age of the fish in days and then multiplied by 1,000 to produce the ceroid index. For purposes of comparison, samples of yellow perch and/or golden shiners were collected from a number of other water bodies.

A water budget for the lake was determined using the following formula:

Outflow + evaporation = precipitation + ground water inflow

Initially outflow values were determined using a rectangular weir at the outlet pipe. After dredging began, the weir was removed to lower the lake level and outflow was determined using lake elevations and Manning's equation to determine flows through the outlet pipe. Rainfall and evaporation were determined from NOAA reports, and groundwater inflow was calculated.

A data storage and retrieval system was developed for this project to facilitate the orderly cataloging and future analysis of the considerable amount of data gathered. In addition several software programs were written to display the data in useful formats or calculate additional parameters. These programs can be made available to the reader upon request.

A novel graphing format was used to allow easy seasonal comparisions for the three and one-half year's data while maintaining a large scale on the graph. Fifteen months' data are shown on each tier of the graph (see Figure 6). The data from the summers (JJA) of 1977 and 1978 are repeated in the lower left of each graph. This repeated sample period is slightly shaded to prevent confusion.

RESULTS

HYDRAULIC DREDGING

Though discussed in more detail in another report (Welch et al. 1979) highlights of the dredging procedures will be summarized here. Dredging occurred from August to November, 1977, and June to August in 1978 using a 'Mudcat' dredge.

A sedimentation basin had been constructed by building dikes at the southeast of the lake (Figure 3). The surface area was 2.42 ha, and it had an average depth of about 2 meters and a usable volume of 48,400 m^3. Dredge effluent was pumped into the basin at the south end, was slowed and evenly distributed by a rubble barrier across the basin, and flowed through an outlet tube into a small wetland immediately adjacent to the lake.

The total solids content of the dredge effluent ranged from 1 to 7 percent solids by weight. Inplace lake sediment was about 30% total solids thus giving a material removal rate of 15-105 m^3/hr. On the average, effluent solids content was about 5%, giving a material removal rate of just under 76 m^3/hr.

The dredge created little or no detectable turbidity. Measurements on July 20, 1977 with a submersible photometer indicated essentially similar light values at comparable depths adjacent to and at distances of 100 m and more from the dredge.

Sediment removal by the settling basin reached 99% and phosphorus removal was also high (Table 1). In the second year of operation, a rapid decrease in detention time occurred as the basin filled. Near the end of dredging, the detention time was estimated at about 1 hour with removal of sediment occurring only in the dispersed channels leading from the dredging influent pipe to the outlet. A value of 1900 mg/l for suspended solids was obtained at the outlet at that time.

After dredging was terminated in 1978, the basin had been filled with material to depths varying from 0.6 m to 3.3 m and contained a volume of 36,900 m^3 of material with a solids content of about 45%. The actual volume of material dredged from the lake was somewhat higher, since the solids content of the inplace lake sediments ranged from 26 to 37%. Using this range of solids content, the removal efficiency of 99.3% and amount of material in the spoils area, we calculated about 56,000 m^3 of sediment had been removed from the lake. This value agrees closely with the 52,000 m^3 calculated from differences in bathymetry.

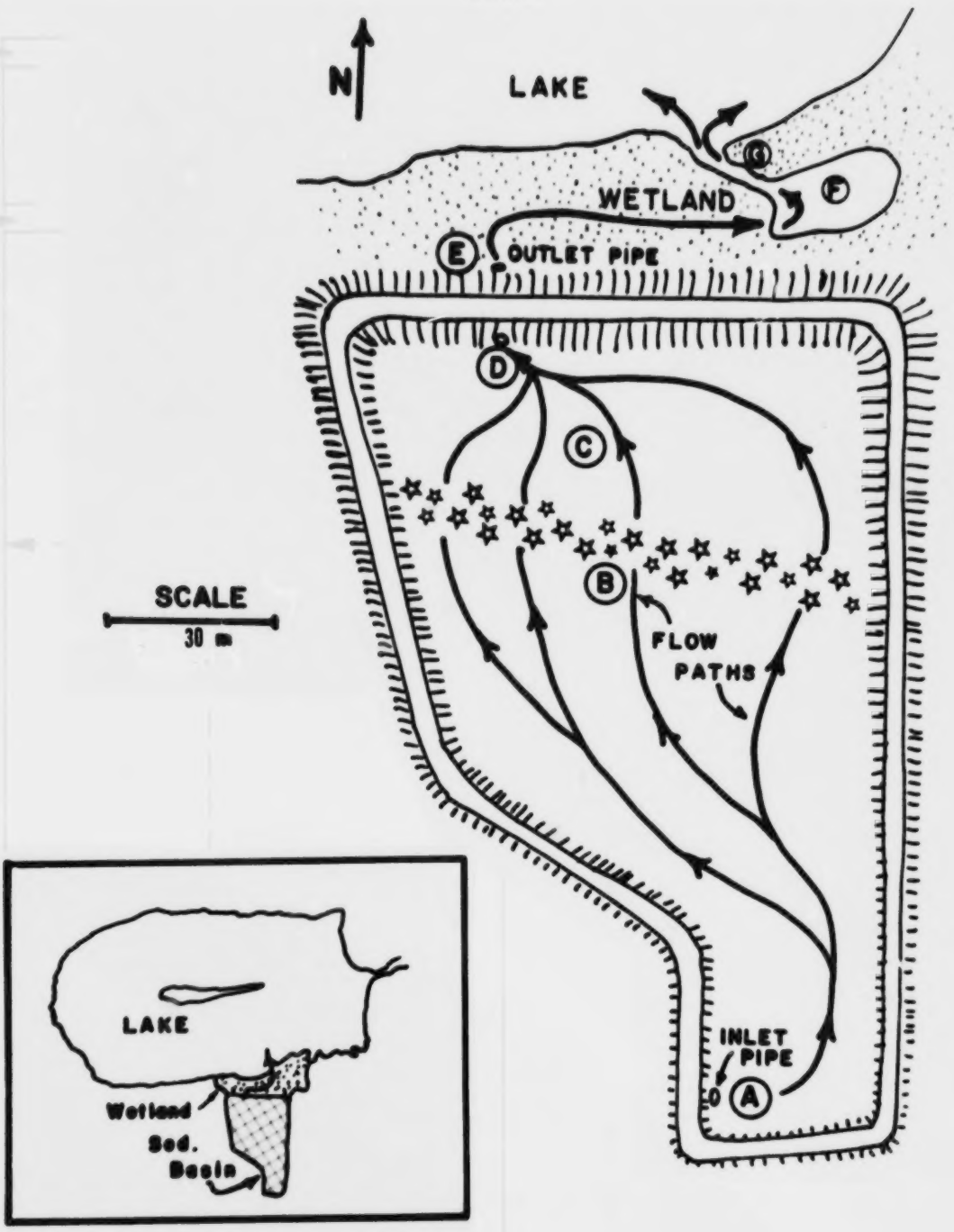


Figure 3. Sedimentation basin for Collins Lake dredged sediments showing flow of water through the basin and back into the lake. Stars represent rubble barrier. Letters indicate sample stations for chemical analysis of the dredged water.

TABLE 1. REMOVAL OF NUTRIENTS AND TURBIDITY BY SEDIMENTATION BASIN IN AUGUST, 1977 (letters refer to Figure 3).

Variable	Location						
	Inlet Pipe A	Concrete Barrier B	Selling Area C	At Outlet E	Seept E	In Wetlands F	Wetlands Outlet G
Tubidity (JTU)	NA	NA	272	388	220	100	55
Suspended Solids (mg/l)	42,285	8.758	184	281	153	20	36
Total Phosphorus (mg/l)	18.11	3.79	1.63	0.206	0.148	0.074	0.076
Soluble Phosphorus (mg/l)	NA	0.051	0.084	0.086	0.083	0.065	0.061
Orthophosphate Phosphorus (mg/l)	NA	0.036	0.055	0.028	0.037	0.033	0.027
Ammonia nitrogen (mg/l)	NA	0.10	0.32	0.16	0.26	0.20	0.16
Nitrate nitrogen (mg/l)	NA	0.13	0.28	0.13	0.37	1.03	0.80

MORPHOMETRY

Initially about 7 ha of lake bottom were scheduled for dredging (Welch et al. 1979). However only 2.6 ha were dredged (Figure 4) so the changes in lake morphometry (Table 2, Figure 5) were not large. During dredging, the lake level was lowered about 0.5 m to facilitate stump removal, and it has only recently (fall 1980) been restored to its original level. The data in Table 2 and Figures 4 and 5 reflect this constant lake level.

WATER BUDGET

In 1976, the yearly precipitation onto the lake was 2.50×10^5 m^3 (103.5 mm) and evaporation from the lake surface was 1.6×10^5 m^3 (64.6 mm). Groundwater input is estimated as 27.6×10^5 m^3/year as based on 7 months of data taken in 1976 (Table 3). The outflow during 1976 is therefore approximately 28.50×10^5 m^3/year or 90.6 liters/sec.

In 1977, the yearly precipitation was 2.90×10^5 m^3 (119 mm) and evaporation was 2.30×10^5 m^3 (96 mm). Groundwater input was 29.9×10^5 m^3/year while the outflow was 30.40×10^5 m^3/year or 96 liters/sec.

In 1978, the precipitation was 2.1×10^5 m^3 (86 mm) and evaporation was 2.7×10^5 m^3 (109 mm). Groundwater input was 29.94×10^5 m^3 while the outflow was 29.4×10^5 m^3 or 93 liters/sec.

Published data for 1979 are insufficient to predict any water budget, but if the previous three years are averaged, then rainfall would be approximately equal to evaporation, and groundwater flow would be approximately equal to lake outflow (93 liters/sec).

With an average yearly groundwater flow rate of 94 liter/sec the nominal detention time of the lake (before dredging) was 82 days. After dredging, with the increase in lake volume, the detention time increased to 92 days.

SEDIMENTS

The sediments in Collins Lake vary from an organic silty clay (muck) to black organic clay in the upper 0.5 to 1 m of the sediment column. Beneath this is a more silty clay with lower organic content, possibly of a fluvial rather than a lacustrine origin (Table 4). There is a marked difference in particle size between the sediments in the deeper western basin and the shallower areas of the lake.

Cores obtained from the shallow eastern part of the lake indicate a textural change at a depth between 8 and 20 cm. Since this area had been a marsh and was flooded when the lake level was raised in the 1940's, this textural change would suggest a long term sedimentation rate Of 0.5 to 0.25 cm/yr. Cores from the undredged areas near South station contain Trapa natans nuts to a depth of 60 cm. Since T. natans was introduced to Collins Lake near the turn of the century, sedimentation rates in this area approach 0.8 cm/yr.

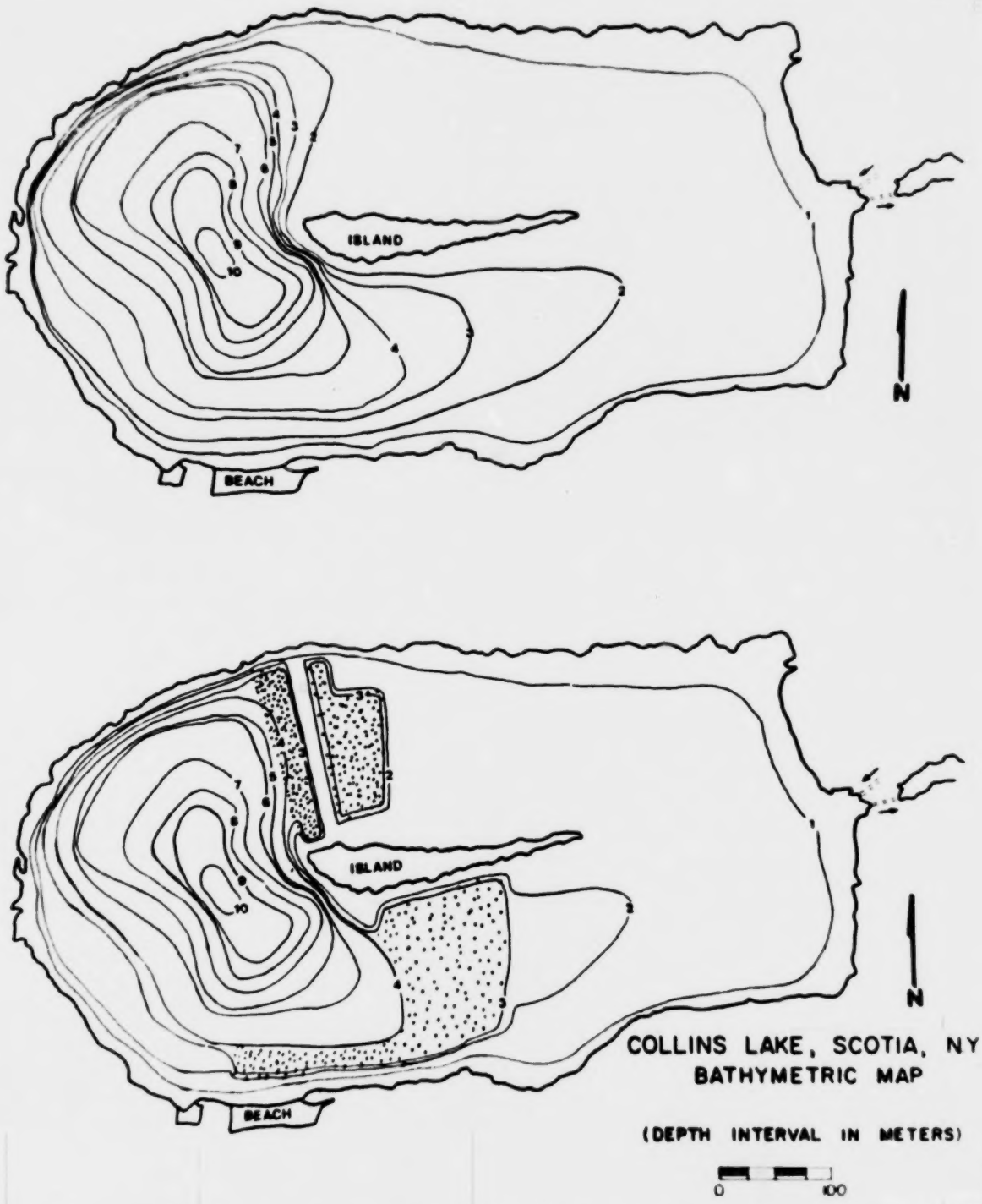


Figure 4. Bathymetry of Collins Lake before (top) and after (bottom) dredging. Stippled areas were dredged.

TABLE 2. MORPHOLOGY OF COLLINS LAKE, SCOTIA, N.Y. BEFORE AND AFTER DREDGING.

Variable	Before	After
Area	24.34 ha	no change
Volume	6.68×10^5 m^3	7.21×10^5 m^3
Mean depth	2.75 m	2.97 m
Maximum depth	8.74 m	no change
Mean depth to thermocline	4.5 m	4.5 m
Typical epilimnetic volume	5.44×10^5 m^3	5.96×10^5 m^3
(% of total)	(81.3%)	(82.6%)
Typical hypolimnetic volume	1.25×10^5 m^3	1.25×10^5 m^3
(% of total)	(18.7%)	(17.3%)
Hydraulic detention time	82.1 days	92.2 days

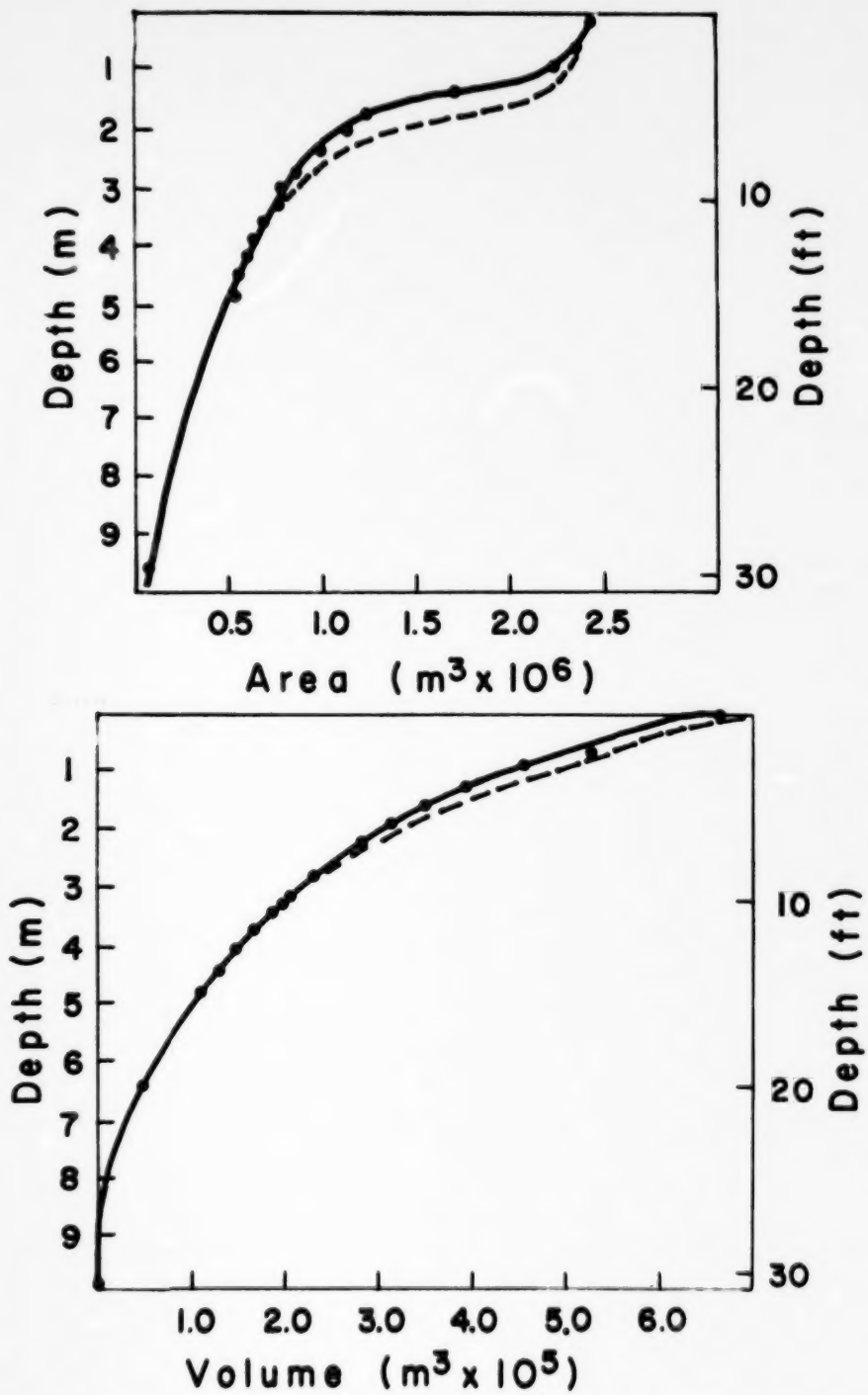


Figure 5. Hypsographic curves of Collins Lake before (solid line) and after (dashed line) dredging.

TABLE 3. WATER BUDGET FOR COLLINS LAKE, SCOTIA, NEW YORK (Outflow was measured. Evaporation and rainfall were based on locally derived data and groundwater inflow was calculated. Surficial runoff, including a storm sewer, are ignored.)

Date Month	Flow out $m^3 \times 10^4$	Evaporation $m^3 \times 10^4$	Rainfall $m^3 \times 10^4$	Groundwater $m^3 \times 10^4$
6/1976	23.85	3.80	3.54	24.12
7	24.27	3.44	2.38	25.33
8	25.03	3.80	3.14	25.70
9	21.29	2.18	2.01	21.45
10	27.30	--	3.43	23.88
11	24.96	--	.82	24.14
12	21.24	--	.80	20.43
1/1977	17.06	--	1.08	15.98
2	13.36	--	1.61	13.20
3	18.20	--	4.11	14.10
4	30.10	3.10[a]	2.39	30.81
5	26.17	4.82	1.35	29.63
6	18.72	3.31	2.11	19.92
		Alcove Reservoir	Schenectady	
7/1977	25.41	4.14	1.21	28.35
8	21.43	3.09	1.87	22.65
9	21.29	2.54	4.25	19.57
10	38.68	1.42[a]	2.91	37.19
11[b]	30.83	.92	3.40	28.35
12	41.72	--	2.73	38.98
1/1978	27.12	--	3.80	23.31
2	8.22	--	.29	7.93
3	20.48	--	1.20	19.28
4	42.58	2.35[a]	1.03[b]	43.89
5	24.65	3.45[a]	1.16[b]	26.94
6	19.82	5.12[a]	2.78	22.17
7	23.89	5.62[a]	3.02	26.49
8	25.03	4.64[c]	2.08	27.59
9/1978	21.29	2.63	1.10	22.82
10	23.51	1.64	2.12	23.03
11	25.69	1.11 (Approx)	.64	26.16
12	31.48	--	1.73[b]	29.74
1/1979	37.17	--	4.22[b]	32.94
2	15.76	--	1.06[b]	14.70
3	19.34	--	1.13[b]	18.21
4	43.68	--	2.41[b]	41.27
5	27.31	--	--	--
6	21.29	--	--	--

		Alcove Reservoir	Schenectady	
7/1977	25.41	4.14	1.21	28.35
8	21.43	3.09	1.87	22.65
9	21.29	2.54	4.25	19.57
10	38.68	1.42[a]	2.91	37.19
11[b]	30.83	.92	3.40	28.35
12	41.72	--	2.73	38.98
1/1978	27.12	--	3.80	23.31
2	8.22	--	.29	7.93
3	20.48	--	1.20	19.28
4	42.58	2.35[a]	1.03[b]	43.89
5	24.65	3.45[a]	1.16[b]	26.94
6	19.82	5.12[a]	2.78	22.17
7	23.89	5.62[a]	3.02	25.49
8	25.03	4.64[c]	2.08	27.59
9/1978	21.29	2.63	1.10	22.82
10	23.51	1.64	2.12	23.03
11	25.69	1.11 (Approx)	.64[b]	26.16
12	31.48	--	1.73[b]	29.74
1/1979	37.17	--	4.22[b]	32.94
2	15.76	--	1.06[b]	14.70
3	19.34	--	1.13[b]	18.21
4	43.68	--	2.41[b]	41.27
5	27.31	--	--	--
6	21.29	--	--	--
7	22.0	--	--	--
8	18.20	--	--	--
Averages	93 liters/sec.	89.9 mm/yr	102.9 mm/yr	90 liters/sec

[a]Data from Geneva, New York.

[b]Data from Albany or Troy, New York.

[c]Data from Valatie, New York.

TABLE 4. SEDIMENT CORE ANALYSES.

Parameter Sediment Type	South Not Dredged Organic Silt & Clay	South Dredged Silt and Clay	West 32' Not Dredged Black Organic Clay
Total-P Interstitial Water	202 µg/l	85 µg/l	6560 µg/l
Ortho-P Interstitial H_2O	76 µg/l	10 µg/l	7025 µg/l
$\%H_2O$ by Weight	72.4%	56.7%	81.1%
%Solids by Weight	27.6%	43.3%	18.9%
%Organics by Weight @550°C	9.6%	6.2%	11.2%
mgP/gm Dry Weight H_2SO_4 and Persulfate Digestion	0.427 mgP/gm	0.438 mgP/gm	0.525 mgP/gm
%Ortho-P in Total-P of Interstitial Water	37%	12%	100%
Grain Size, mm (± one standard deviation)	0.025 ± .08		0.01 ± .016
%Sand	30-50%		3-20%
%Silt	30-60%		50-65%
%Clay	10-20%		15-35%

In the dredged areas at South Station where almost 2 meters of sediment had been removed, only a silty clay with no organic debris was found. However, the very top of the cores in the dredged areas at South Station had black, highly organic layers at the top. This was attributed to the slumping of organic sediments from the undredged areas into adjacent dredged areas.

Heavy Metals

In February 1977, prior to dredging, iron, copper and lead concentrations were determined for sediments collected from three sites. Interstitial water contained 3.7 to 15.0 mg/l iron, less than 10 µg/l lead and less than 6 µg/l copper. Heavy metals released by acid washing of the sediments ranged from 29 to 40 mg/kg iron, 140 to 220 µg/kg lead and 43 to 63 µg/kg copper.

To simulate the influence of dredging on the mobility of these three metals, sediments from station South were mixed in a blender to duplicate pump turbulence and then analyzed. Water samples were taken after one and ten days, the longer period approximating the initial retention time in the sedimentation basin. After one day iron levels were 5.7 mg/l and after ten days 0.09 mg/l. Similarly, lead concentration declined from 19 µg/l after one day to less than 10 µg/l after ten days and copper concentrations were 770 µg/l after one day and less than 50 µg/l after ten days. In each case the ten day old water demonstrated concentrations at or below the initial interstitial water concentration. One day old water had iron concentrations about that of the interstitial water. Lead was slightly higher and copper was much higher than interstitial values after one day.

X-ray fluorescence studies provided information on the concentration of various metals in the sediments (Table 5).

TABLE 5. CONCENTRATIONS OF METALS IN SEDIMENT SAMPLES (as determined by X-ray fluorescence on cores sampled at 50 cm below the sediment-water interface).

Metal	Concentration (ppm) for five sites				
Iron	28,000	47,000	44,000	44,000	35,000
Nickel	20	53	37	33	39
Copper	54	21	14	64	43
Zinc	150	100	84	160	160
Titanium	2,200	4,000	3,600	2,500	2,300
Vanadium	110	190	180	90	140

In summary, iron was most prominent but typical of many such lakes. The levels of lead were not unusually high in the lake basin. Copper was evident but not at a level to have an influence on phytoplankton. Titanium and vanadium were remarkably abundant but were probably immobile, being tied up in clay minerals (Biggs and Swinehart 1976).

Metals in Fish

Mercury levels were determined for the white sucker, Catostumus commersonni. Twenty specimens were composited to yield a concentration of 0.11 ppm, a value within the range encountered for most fish of the region.

Polychlorinated Biphenyls

Because of several regional problems with polychlorinated biphenyls, PCBs were evaluated in samples collected in November, 1977. Water levels were 2 ng/l and sediment within the settling basin contained 4 µg/kg. These values fall within the usual ranges found in uncontaminated systems.

WATER CHEMISTRY

In general Collins Lake is typical of a moderately hard water, productive, thermally stratified lake (Table 6). Bottom waters during the summer exhibit elevated levels of orthophosphate phosphorus and ammonia nitrogen. The north shore springs, which contribute 95% of the water input to Collins Lake, are high in nitrate nitrogen and low in phosphorus.

Temperature and Dissolved Oxygen

Collins Lake is a dimictic lake with ice cover during the winter months (Figure 6). During the summer, the lake is thermally stratified with an anoxic hypolimnion (Figure 7). Thermal stratification at West Station develops in mid-May after the ice melts (Figure 6). The average thermocline depth is about 4 meters and summer epilimnetic temperatures vary between 25° and 30°C.

The variability of dissolved oxygen, shown in Figure 8 as the percent saturation of dissolved oxygen, can be correlated to the biological dynamics within the lake, especially in the shallow (0.5 m) North and South Stations. In these sections of the lake, supersaturation occurs from July through September when algal populations are high. Severe depletion (20% saturation) occurs during the winter under ice cover. Effects of dredging on O_2 saturation can be seen by comparing the dredged South Station to the undredged North Station (Figure 8). In June of 1978 and 1979, undersaturated O_2 values were found at the undredged North Station and not at the dredged South. P. crispus was not present in this dredged area.

TABLE 6. MAXIMUM AND MINIMUM VALUES FOR CHEMICAL PARAMETERS.

Parameters	West (.5 m)	West (8 m)	Groundwater
pH	7.0 - 8.6	6.6 - 7.2	7.3 - 7.6
Alkalinity mg/l as $CaCO_3$	130 - 170	180 - 220	190 - 220
Acidity mg/l as CO_2	0 - 3	3 - 25	6 - 15
Conductivity micromhos/cm	300 - 390	330 - 450	370 - 500
Hardness mg/l as $CaCO_3$	130 - 190	150 - 190	200 - 240
Calcium mg/l	35 - 55	50 - 70	50 - 70
Magnesium mg/l	8 - 15	10 - 25	12 - 20
Temperature °C	0 - 28	4 - 11	7.0 - 10.5
Dissolved Oxygen mg/l	6.5 - 14	0 - 11	6 - 11
PO_4-P (Ortho) µg/l	2 - 40	10 - 550	0 - 5
Total - P µg/l	15 - 56	25 - 550	2 - 10
NO_3-N mg/l	0.1 - 1.0	0.2 - 1.1	0.5 - 2
NH_3-N mg/l	0.1 - 1.0	0.2 - .6	0.1 - .4
Flow	51 to 195 liters/sec		Average of 91 liters/sec

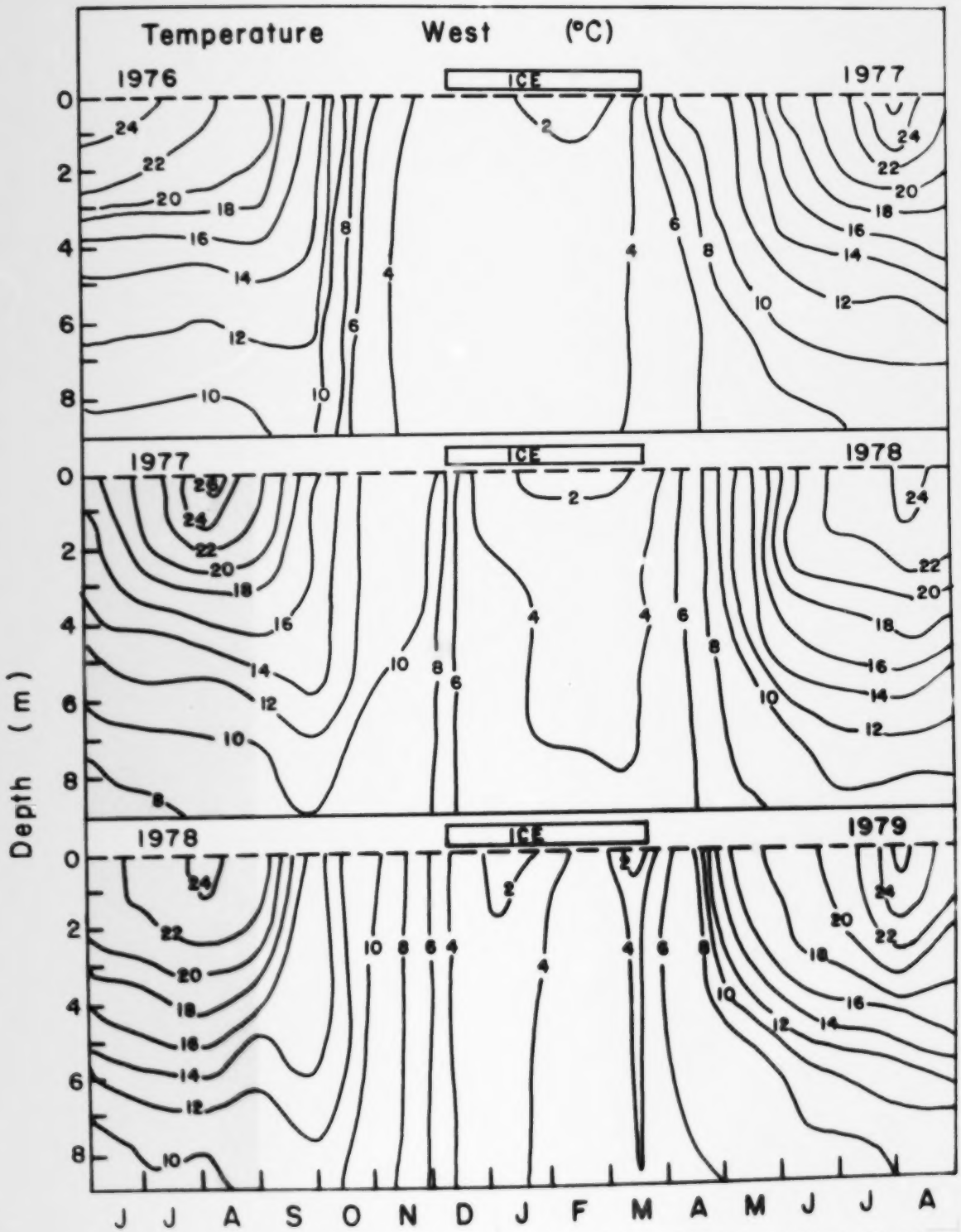


Figure 6. Temperature isopleths versus depth and time at West Station.

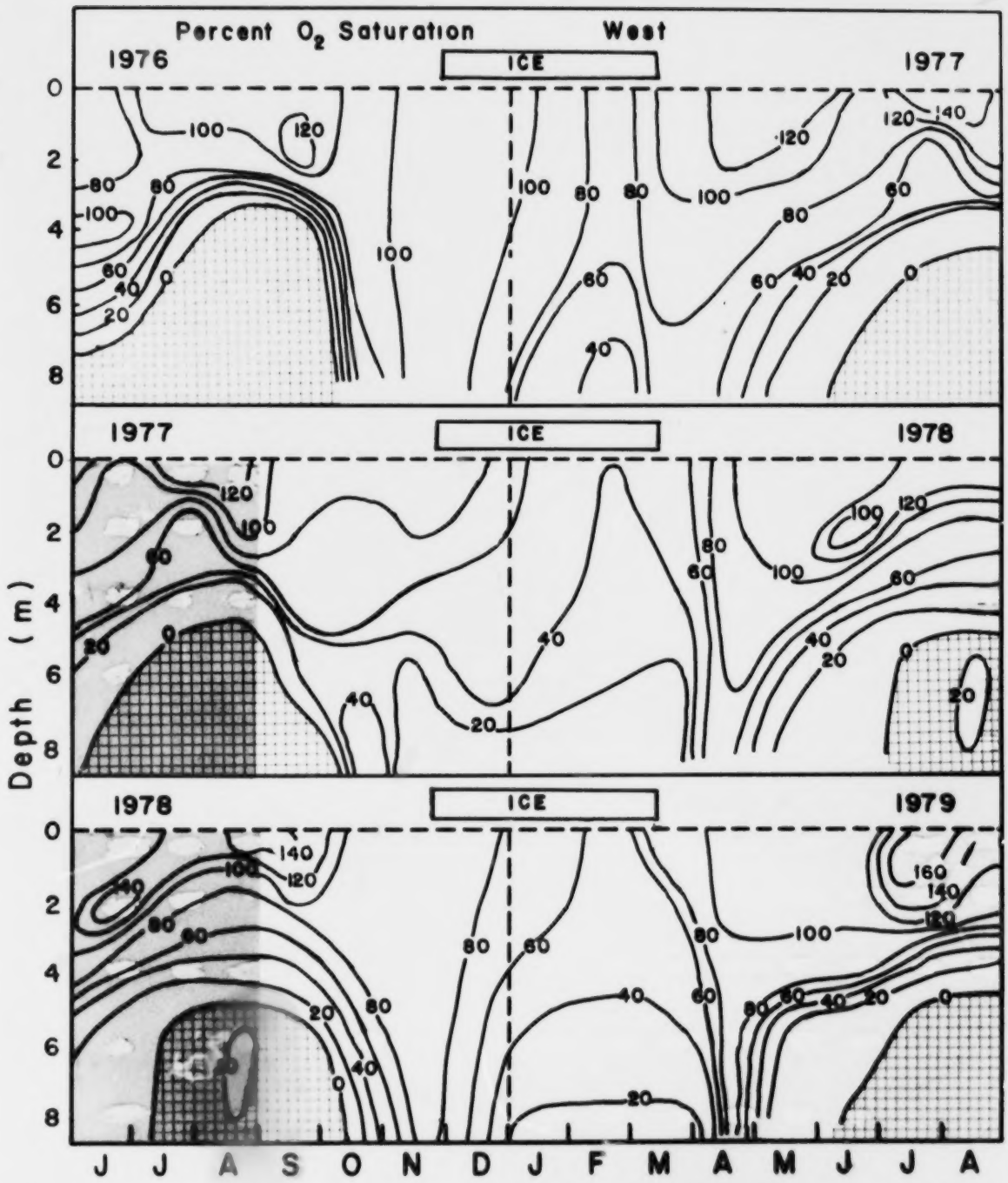


Figure 7. Percent saturation of dissolved oxygen isopleths at West Station. Cross-hatched area represents anoxic conditions.

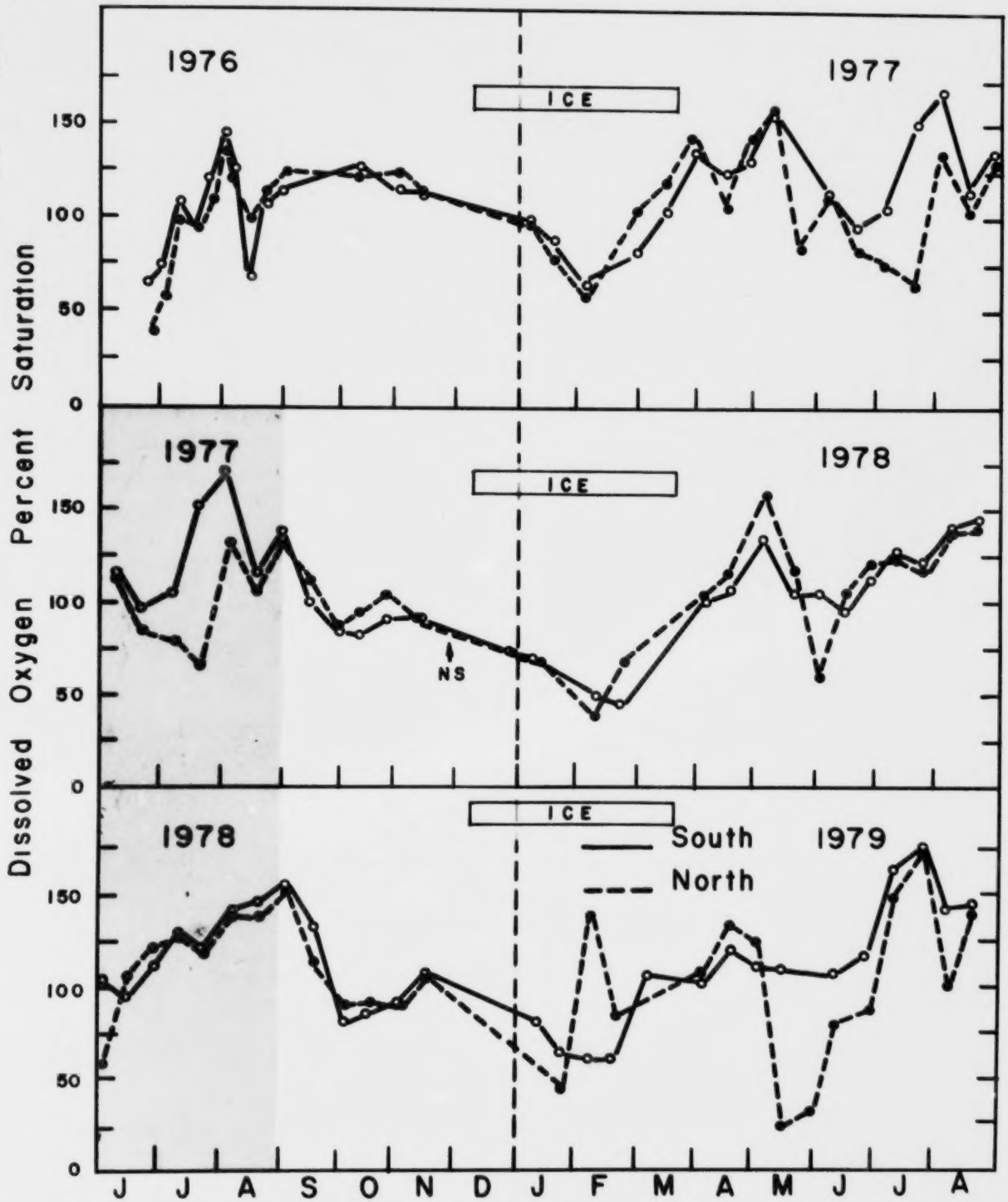


Figure 8. Percent saturation of dissolved oxygen (0.5 m) versus time at North and South Stations. South Stations was moved at "NS" into the dredged portion of the lake. North Station remained undredged.

pH, Alkalinity, Hardness, Conductivity

Values of pH (Figure 9) at West Station are higher from late June to early September (7.9 to 8.7). Higher values (9.1 to 9.6) occur in late May and early June in the shallow, undredged areas. East and North Stations, undredged and shallow, consistently had higher pH values (9.1 to 9.8) during this time than did South Station (8.4 to 8.6).

Conversely, during winter ice cover, pH was lower (6.5 to 7.3). Also, in almost all cases, an abrupt drop in pH occurred in late June at East and North (undredged) Stations. This drop did not occur at South Station (dredged) in 1979.

Patterns and concentration of alkalinity, hardness and conductivity found in Collins Lake were generally characteristic of a moderately hard water, productive lake with an anoxic hypolimnion. No significant differences were found in these parameters between dredged and undredged areas.

Nitrogen

Ammonia (NH_3-N) concentrations at the two shallow stations (Figure 10) were similar during non-ice cover periods but, during the winter, South Station had slightly higher concentration. Ammonia concentrations at West Station were highest (5 mg/l) in the hypolimnion during anoxic summer periods (Figure 11). Epilimnetic values were much lower (0.5 mg/l) during the summer and were similar to the shallow stations.

The annual variations of nitrate (NO_3-N) at the two shallow stations (Figure 12) show a maximum under the ice, a rapid decrease after ice melt and a gradual decrease through the summer and fall. Late fall was a period of increasing values. North Station had higher concentrations of NO_3-N during the winter period which may be due to its proximity to high nitrate goundwater input. Seasonal variations in the epiliminion of West Station were similar to those of the shallow stations (Figure 13). Spatial and temporal variations in both ammonia and nitrate appear to show no apparent differences due to dredging.

Total Kjeldahl nitrogen (TKN) values typically range between 0.3 and 0.7 mg N/l, with higher concentrations to 4.14 mg N/l found in the anoxic hypolimnion of the deep West Station during the late spring and summer (Table 7). These high values are undoubtedly due to an increase in ammonia at this time (Figure 11). The range of epilimnetic TKN values found in Collins Lake are comparable the lower values listed for other lakes by Wetzel (1975).

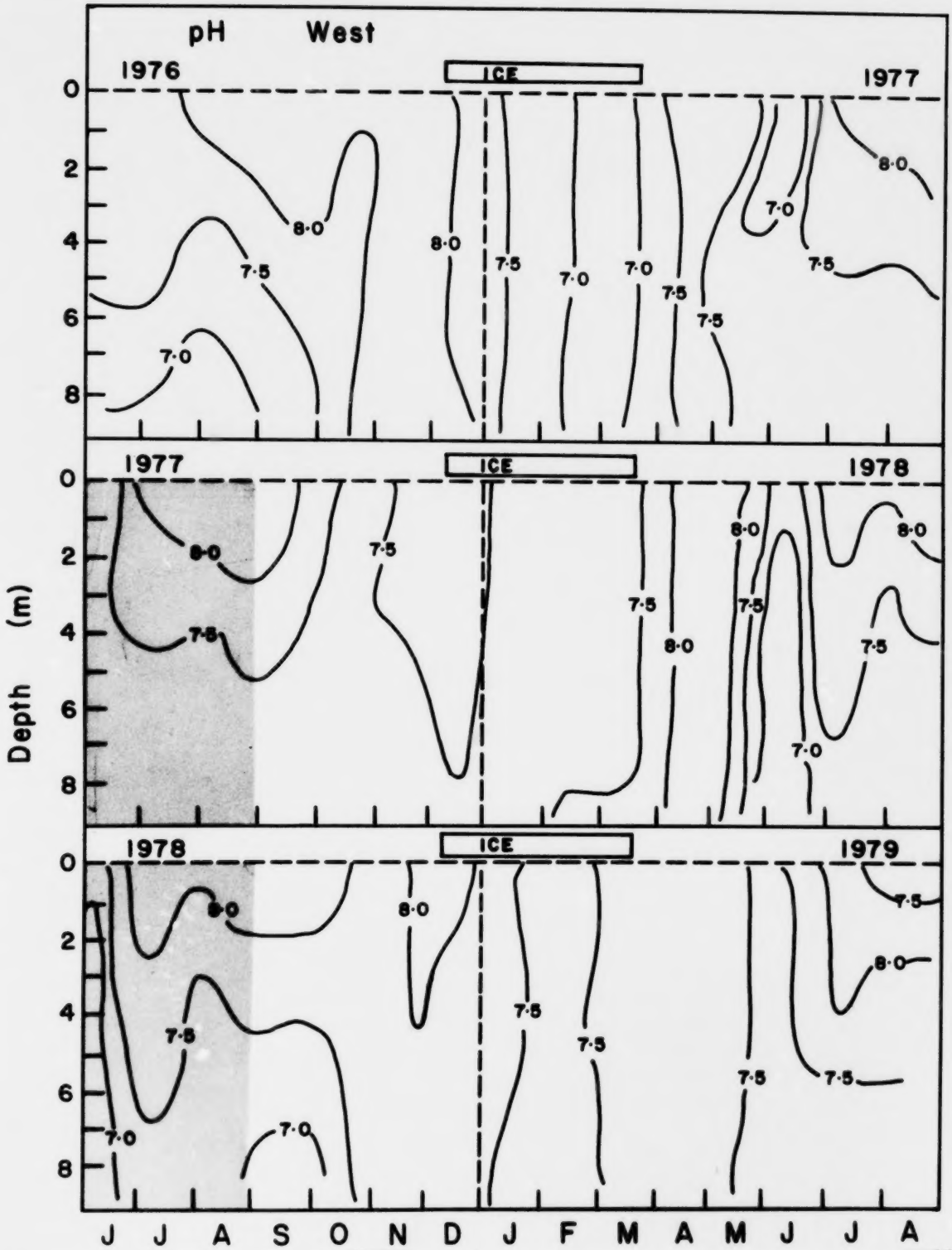


Figure 9. Isopleths of pH at West Station.

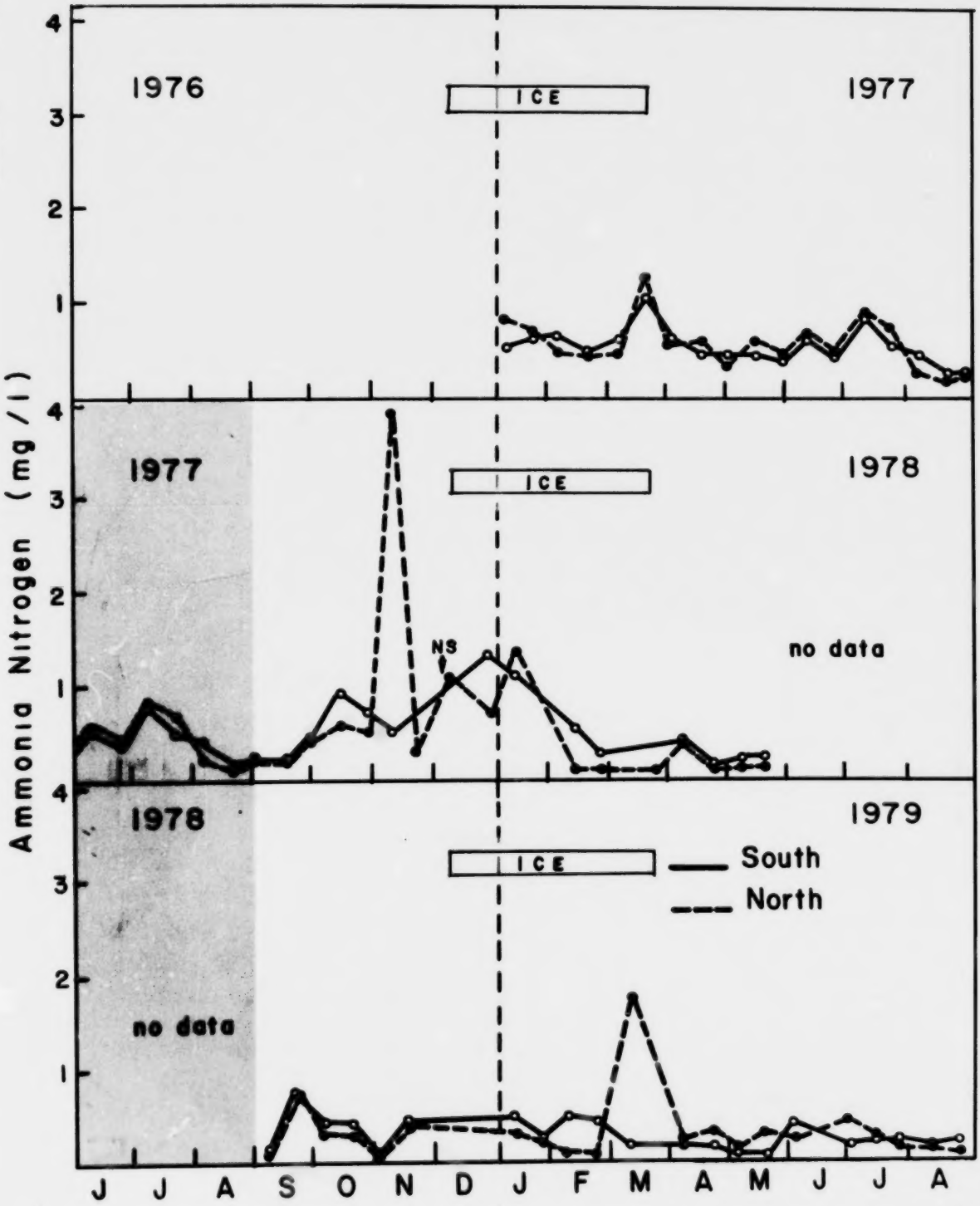


Figure 10. Ammonia (NH_3-N, mg/l) at 0.5 m versus time for North and South Stations. South Station moved at "NS".

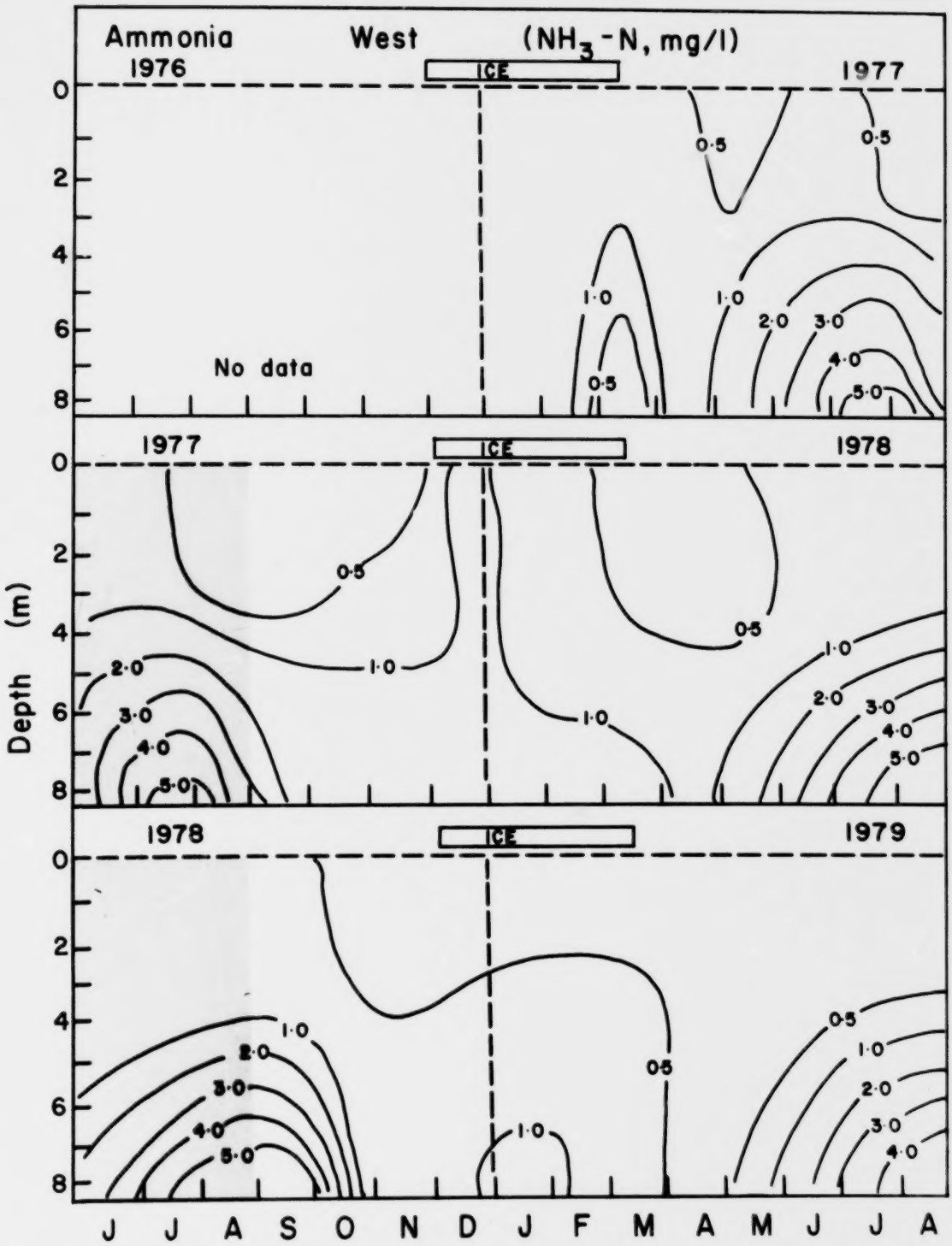


Figure 11. Ammonia (NH_3-N, mg/l) isopleths at West Station.

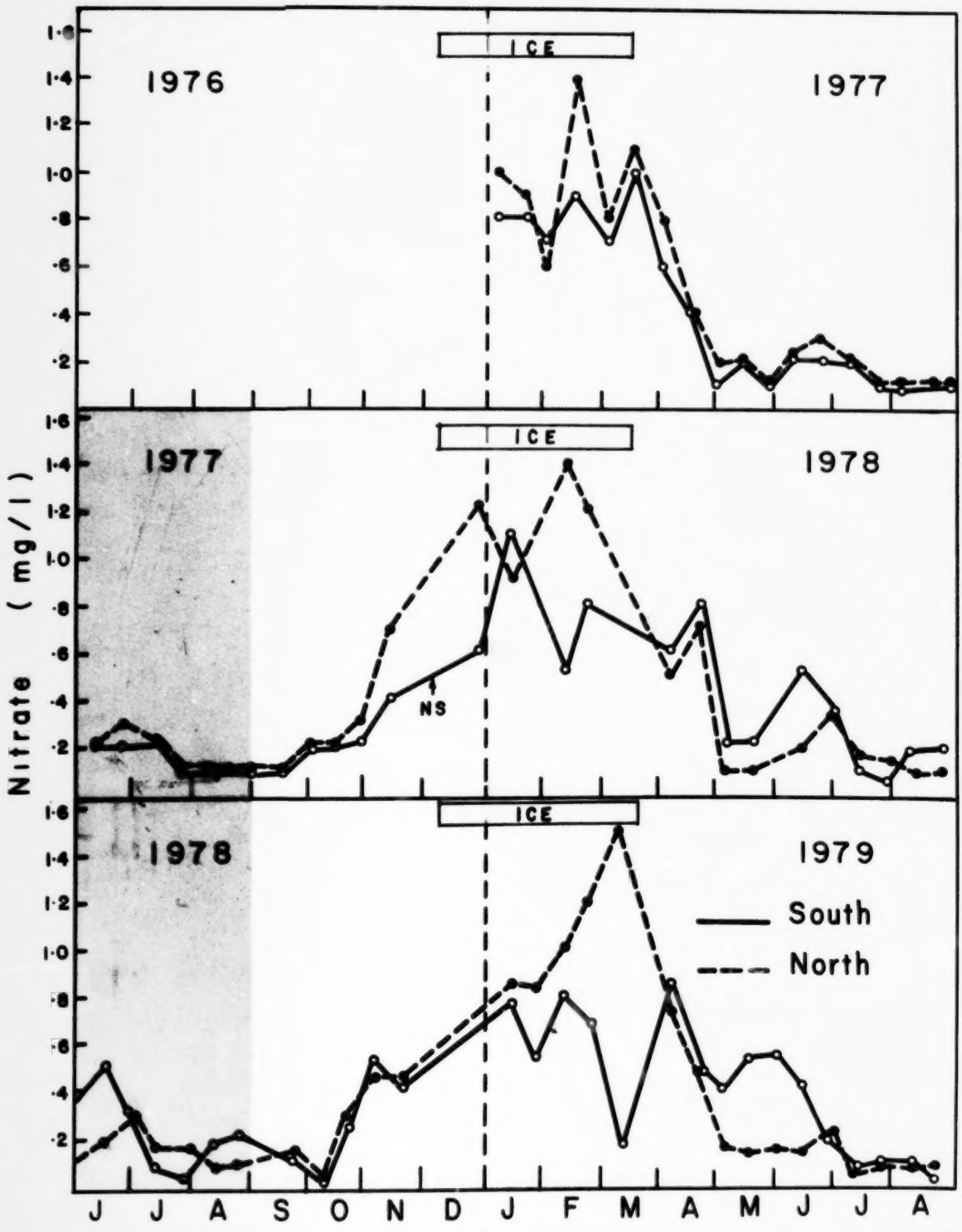


Figure 12. Nitrate (NO_3-N, mg/l) at 0.5 m versus time at North and South Stations. South Station was moved at "NS".

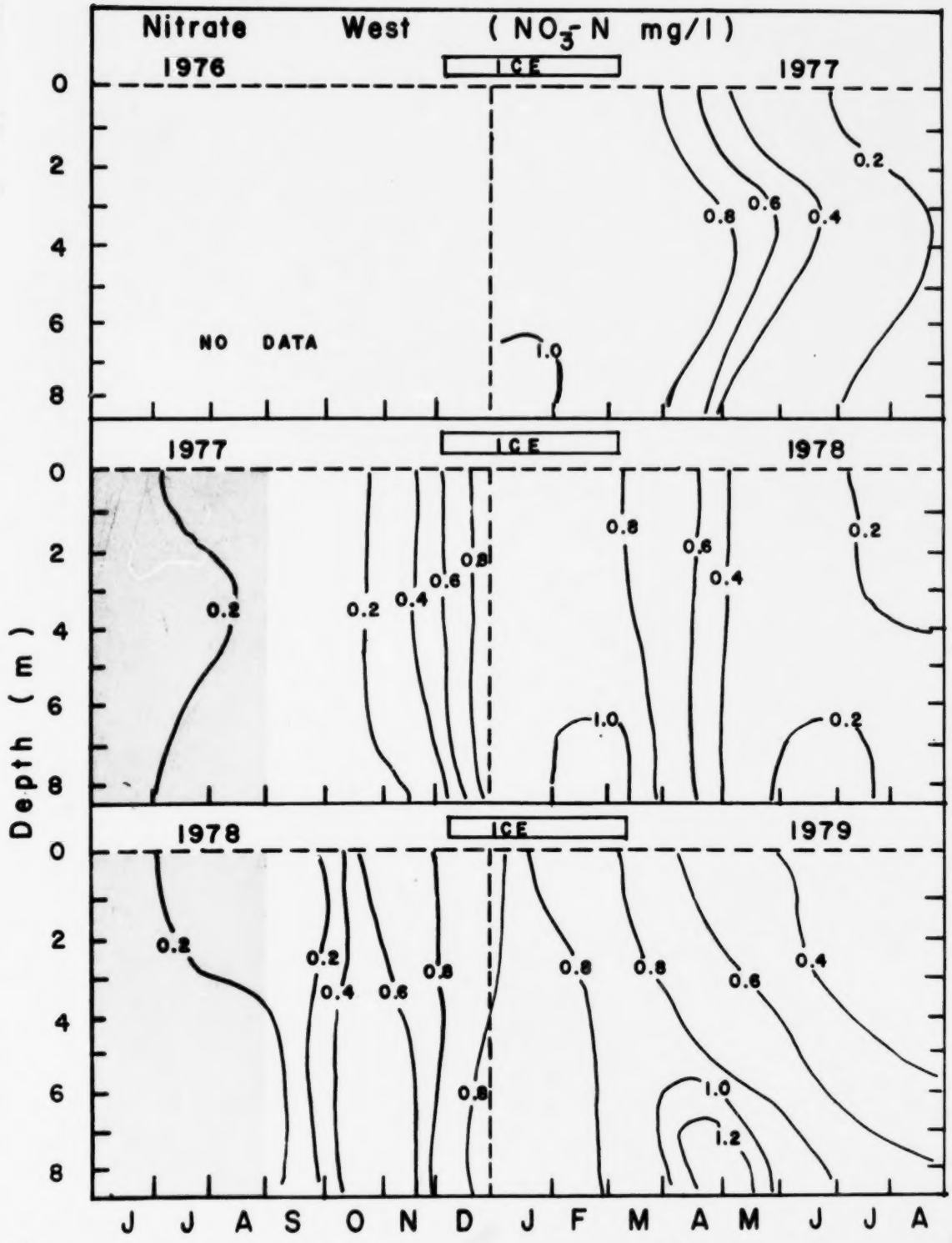


Figure 13. Nitrate (NO_3-N, mg/l) isopleths at West Station.

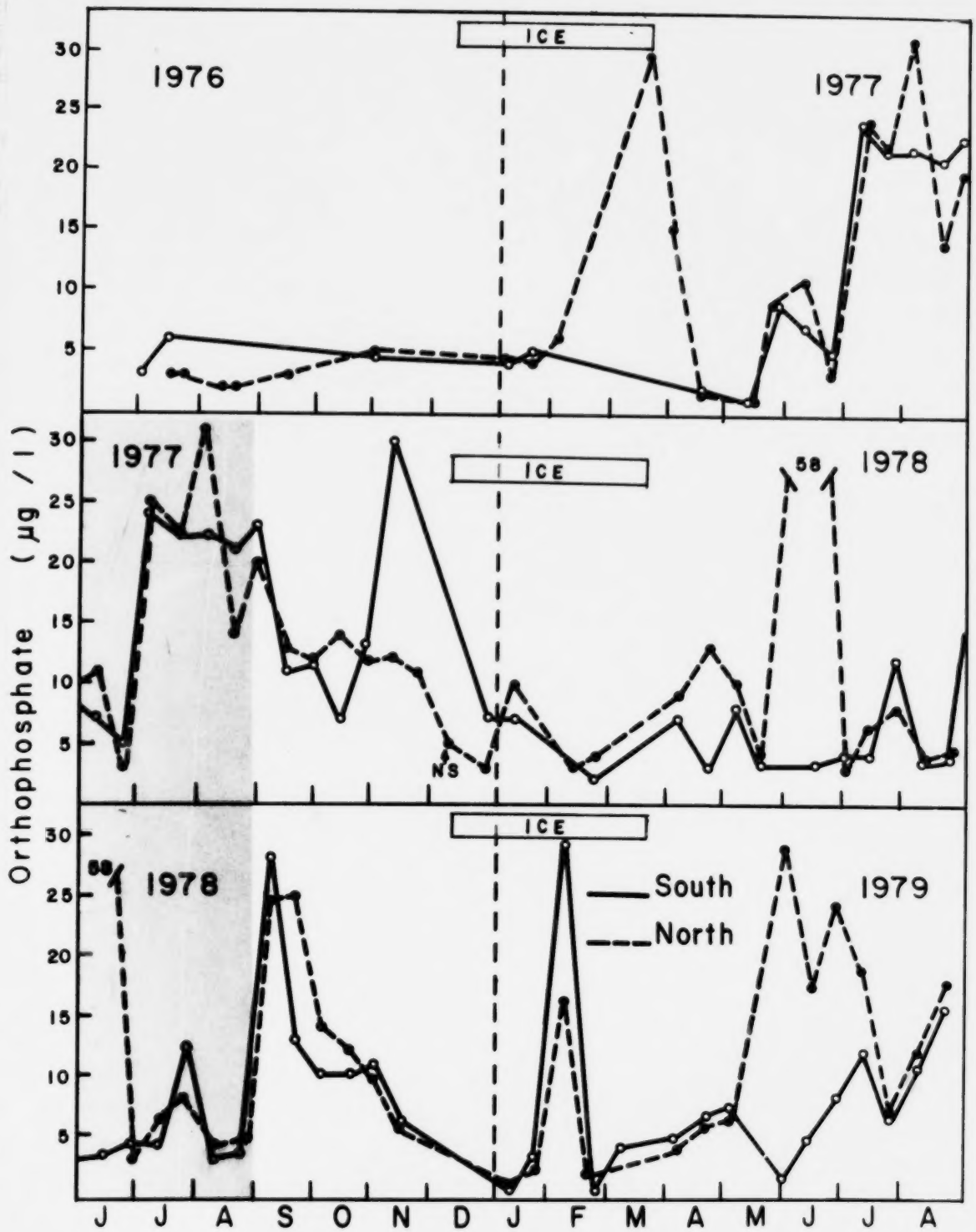


Figure 14. Orthophosphate (PO_4-P, µg/l) at 0.5 m versus time at North and South Stations. South Station moved at "NS".

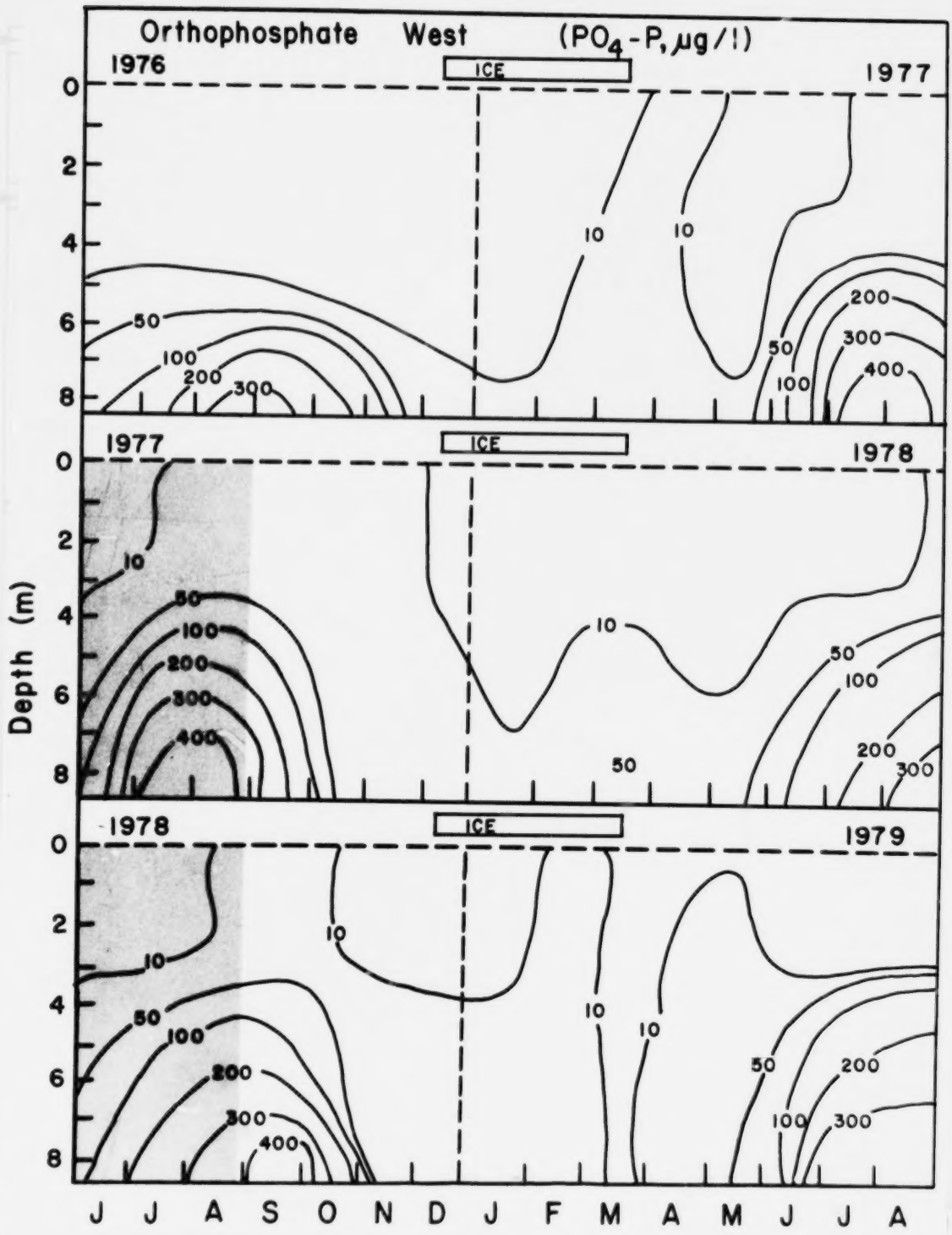


Figure 15. Orthophosphate (PO_4-P, µg/l) isopleths at West Station.

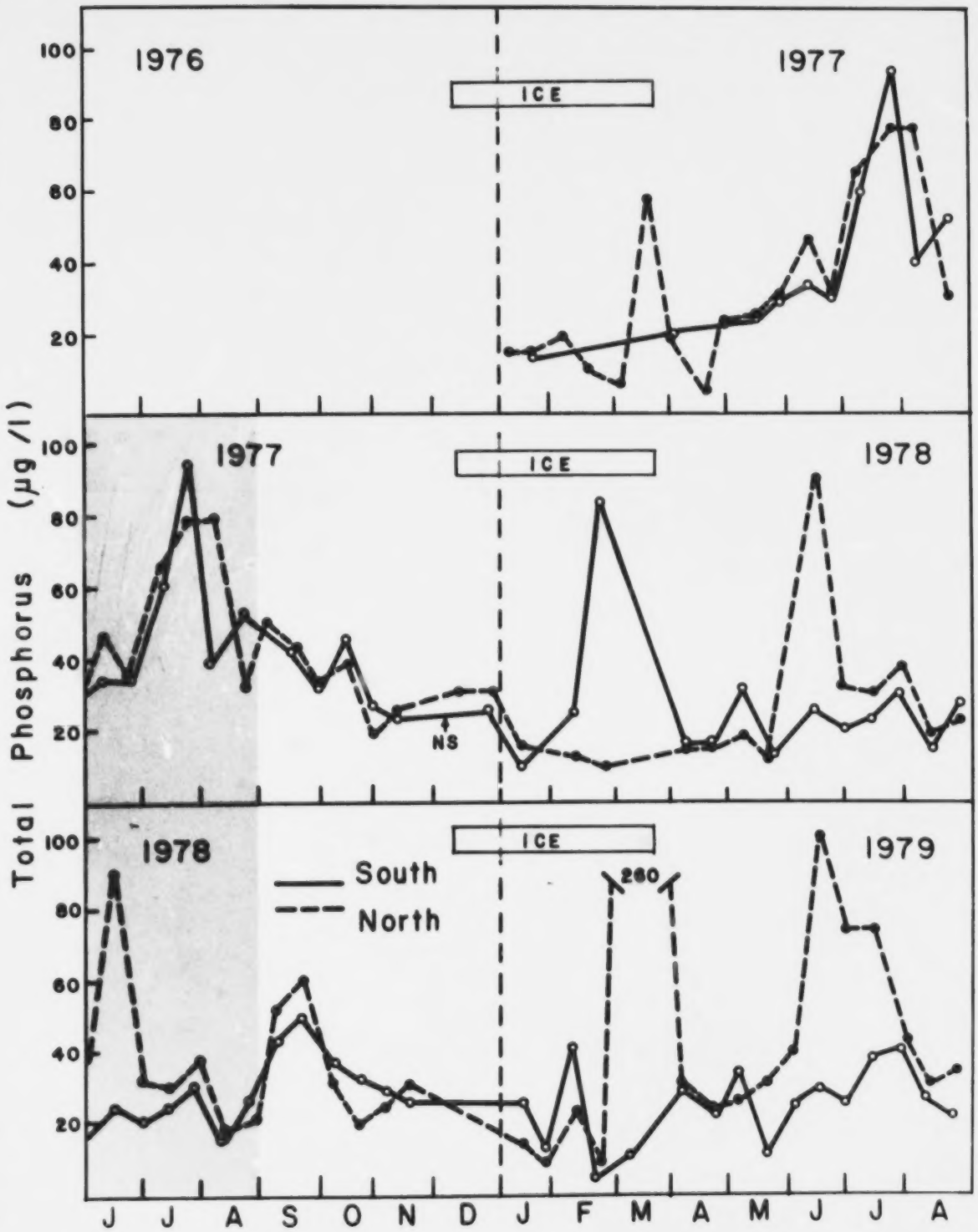


Figure 16. Total phosphorus (µg/l as P) at 0.5 m at North and South Stations versus time. South Station was moved at "NS".

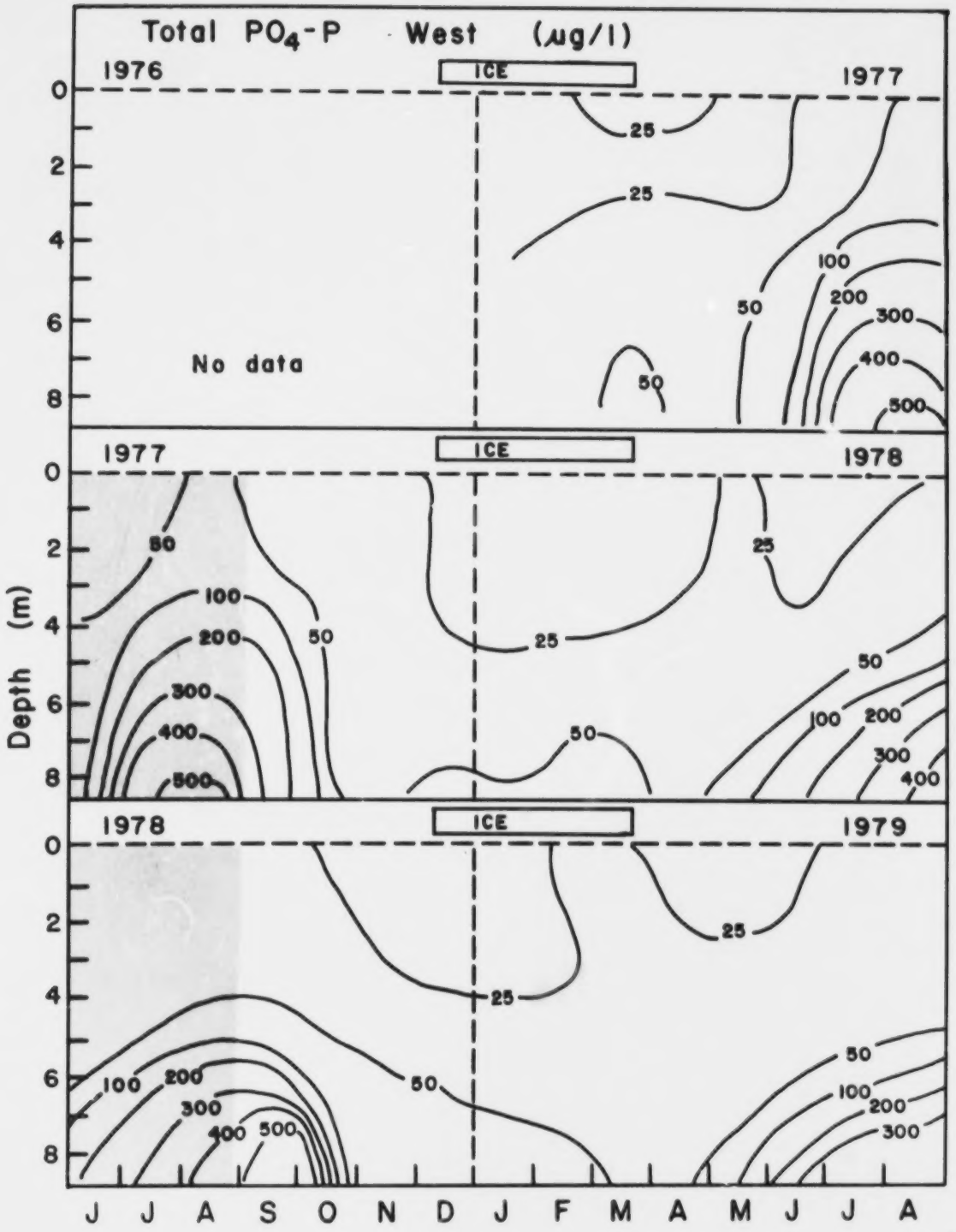


Figure 17. Total phosphorus (µg/l as P) isopleths at West Station.

stemmed. At depths less than 0.5 m P. crispus is also sparse, rarely emerging to the surface, and it has never been seen to flower here. However, plants are present in this zone throughout the summer. This shallow zone which is devoid of vigorous P. crispus growth is probably the result of winter ice cover. Other plants would not be susceptible to ice damage since they are not active during the winter. The area of the lake between the depths of 0.5 and 2.0 m is considerable (Figure 18), and in this region P. crispus forms a virtual monoculture with surprisingly similar densities and biomass.

In Collins Lake P. crispus reproduces mainly by means of modified vegetative apices called turions. They were always present in our quantitative samples and reached densities of over 200 m^2 in some places. Turions germinate in late August, and the young plants proliferate and grow to lengths of about 15 cm by mid November. Densities of over 100 plants per m^2 are present in some patches. They overwinter in this condition under the ice and grow to about 20 cm by January. Between November and April plant density increases are most likely due to rhizome proliferation.

Rapid apical growth occurs in the spring. In '978 plants grew 3.8 cm/day between 27 April and 14 May, and 7.2 cm/day between 24 April and 3 May in 1979. Biomass accumulation is most rapid at this time (Figure 19).

In May approximately 60% of the lake surface is covered with P. crispus floating apices (Figure 18). During late May, the adaxial (upper) surface of the leaves become covered with a layer of $CaCO_3$.

After seed set in early June, the plants undergo a rapid senescence. Plants lose their color and vigor, and most are attacked by chironomid larvae which strip the leaves of their lamina, leaving only the midrib. This is a very rapid process, taking only a week or two in early to mid June (Woodard 1978). The plants both with and without stripped leaves, lose their buoyancy and sink to the bottom where they quickly decompose. By mid July there is no measurable biomass of P. crispus in the entire area where it had been so prevalent. The cycle described for Collins Lake roughly coincides with that recorded by Wehrmeister (1978) for plants in Ohio.

Dredging Effect on P. crispus

There was no observed change in the life cycle as described above during the period of the study. However, the maximum biomass/m^2 in the region of the lake where P. crispus thrives (0.5 m to 2.0 m) apparently increased (Figure 19). The cause of this increase is not known, but the lowering of the lake level by 0.5 m would allow more light to penetrate the water column.

Inside the dredged areas, P. crispus grew very poorly (Table 8). Plants were very infrequent and quite spindly, with very long internodes. They only rarely reached the surface, and they were never seen to flower. Some plants, however, did form turions.

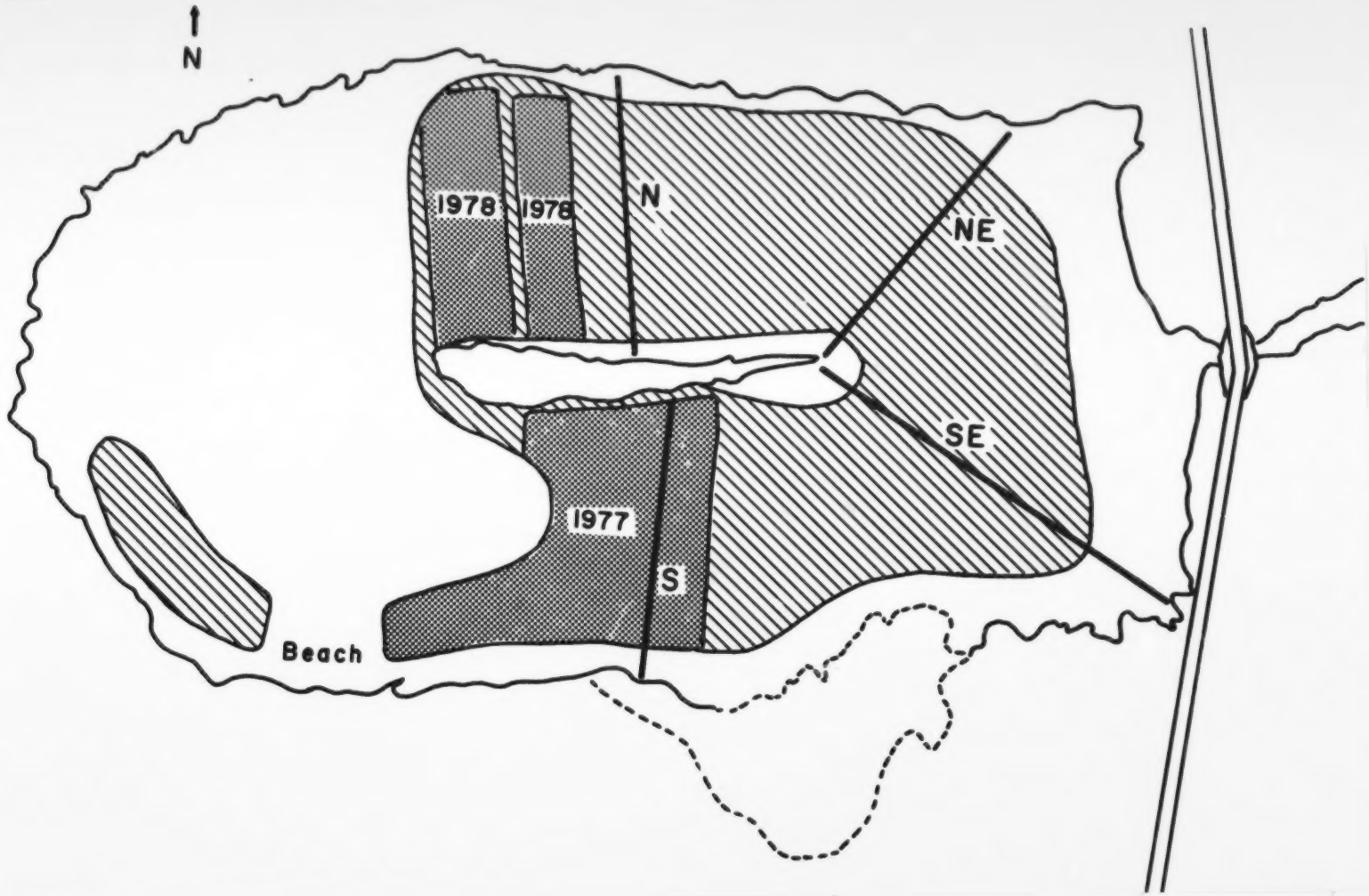


Figure 18. Map of P. crispus extent (stippled and hatching), dredged areas stippled) and transects (lines).

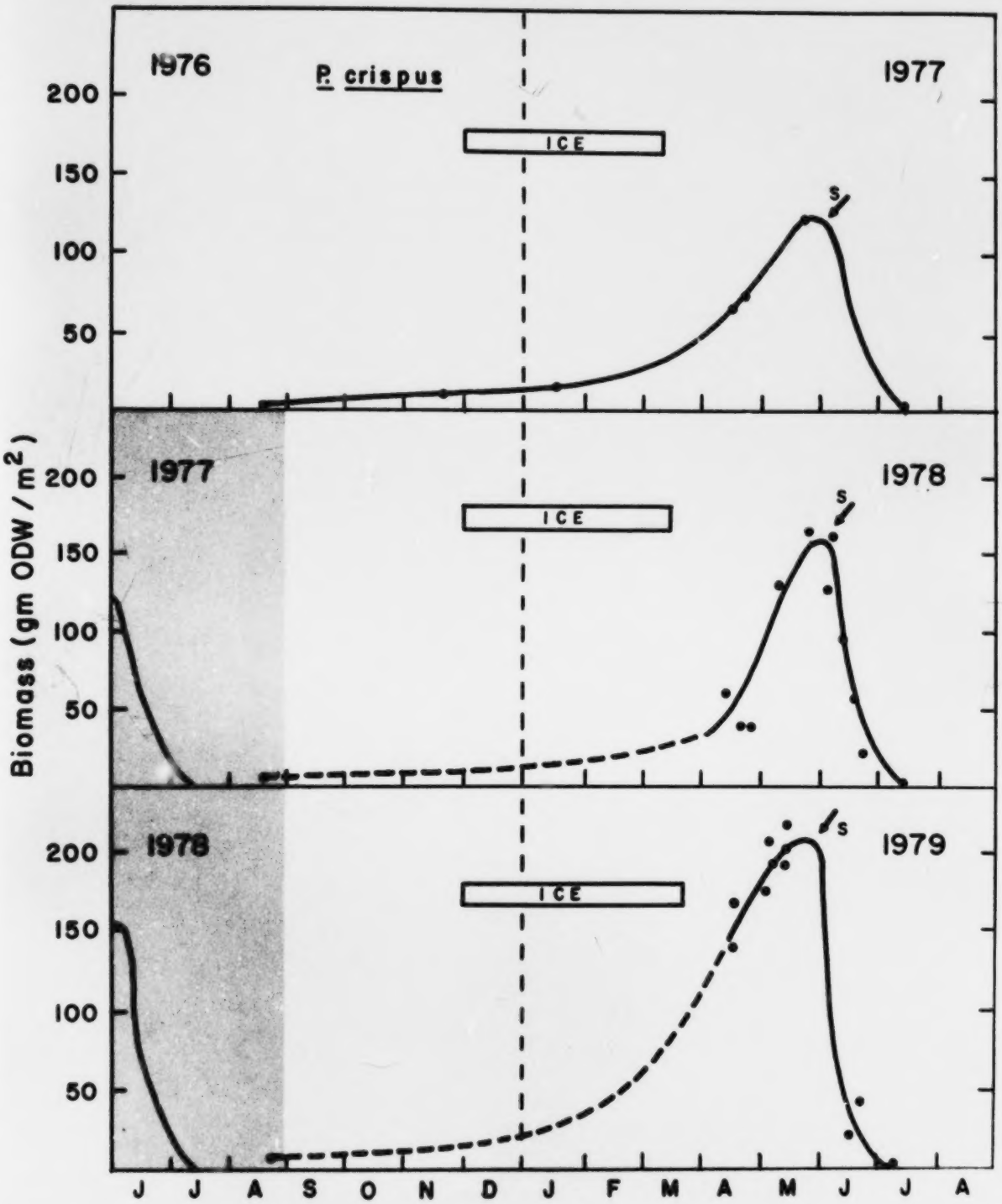


Figure 19. Biomass of P. crispus versus time. Broken line indicates interpolated data, "s" indicates senscence.

TABLE 8. P. CRISPUS BIOMASS ESTIMATES IN DREDGED AND NON-DREDGED AREAS (Samples were taken with (a) cylindrical sampler, (b) rake alone, and (c) diver sampling quadrats; all data are from 1979, all values in gm ODW/m^2).

	Area Dredged		
Sample Date	1977 (after 2 years)	1978 (after 1 year)	Nearby Undredged
22 April	0 (b)	Not sampled	App. 150 (a)
3 May	0.2 (b)	0.2 (b)	180 (a)
8 May	0.3 (b)	0.4 (b)	200 (a)
15 May	0.3 (c)	0.3 (c)	215 (c)

In the dredged areas plant densities of 3-4/m^2 were typical, though distribution was spotty. At the same time, densities in nearby undredged areas were 800-1000/m^2. Most of the plants in the dredged areas were 30-50 cm long in mid May when the plants from nearby undredged areas were 1.5 m. Plants growing on the organic material slumped into the dredged areas from immediately adjacent undredged areas were longer and more vigorous, and they grew at greater densities than plants from other parts of the dredged area.

In mid May of 1979, plant densities, vigor and lengths appeared to be greater in the area dredged one year before (1978) than in the area dredged two years before (1977). This observation was reinforced by notes taken during a 1978 dive into the then-recent 1977 dredged area south of the island. Both these notes and photographs taken at that time demonstrate a denser and more vigorous P. crispus population south of the island in 1978 than was found in 1979. In summary, it appears that P. crispus does not rapidly invade the newly dredged areas, but rather its success in these areas decreases with time. The decrease in P. crispus in the dredged areas is not merely a function of propagule density, but must be related to other more long-lived parameters. To this date, dredging has been a very effective way of controlling the P. crispus populations in Collins Lake.

Trapa natans

Trapa natans, a Eurasian import, was introduced to the lake in about 1880. It has a large, caltrop-shaped fruit containing barbed spines on each point. It germinates in early summer and rapidly elongates its stem toward the surface. On the stem develop some filiform submerged leaf-like stipules and a terminal rosette of leaves with inflated petioles for buoyancy. These stems can be quite long (over 4 m) in Collins Lake. Floating rosettes appear in late June, flowers in early July, and fruits shortly thereafter.

In recent years, T. natans has been a constant element in the flora of Collins Lake, maintained at a low level through annual hand-pulling of a few

bushels. The lake sediment contains millions of T. natans fruits, most of which are remnants of the past decades when T. natans dominated the lake. During the dredging, many of these fruits were dislodged from the sediment, escaped the suction of dredge and were blown by the prevailing wind eastward. All of these floating fruits that we examined were hollow.

In the summer of 1978, there was a dramatic resurgence of T. natans to the south of the island in both the newly dredged area and to the east of it. At least 10,000 plants were hand-picked at that time, but not before many had fruits. The outbreak in 1979 was significant (perhaps several hundred plants), but was not of the magnitude of the 1978 population. This resurgence of T. natans in Collins Lake was totally unpredicted. The distribution of the resurgent T. natans population suggested that it could have been caused by the dredging, but other factors could have been involved.

Other Plants

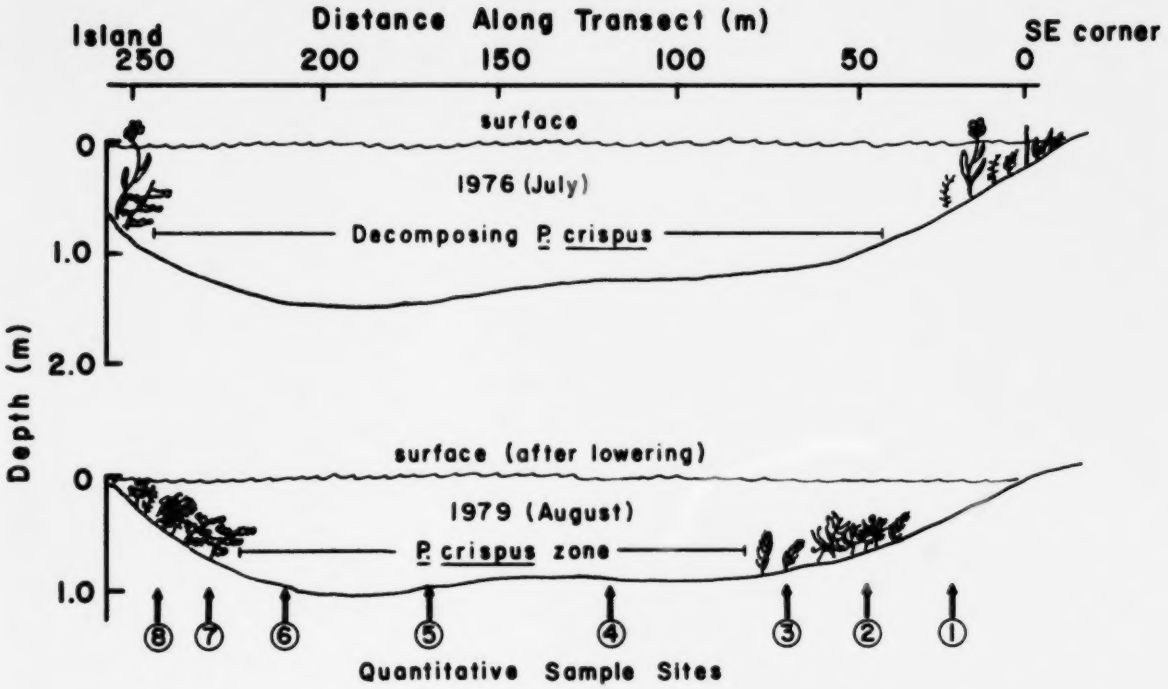
Another aquatic plant, Najas minor, also showed a marked increase in dominance in the lake during the study. The portions of the lake shallower than 0.5 m were not dominated by P. crispus and contained a diverse community of littoral plants including Anacharis, Ceratophyllum, Heteranthera, Nymphaea, Potamogeton pectinatus and N. minor, among others. In the plant surveys prior to dredging (Table 9), N. minor was not an important element in this community. After the lake level was lowered in 1977, the P. crispus-free zone was moved toward the center of the lake. This created a new habitat for this shallow water community. During the summer of 1979, N. minor dominated large regions of this new habitat in virtually pure stands, reaching biomass in excess of 300 gm ODW/m^2 (Figure 20).

The lowering of the lake level had a profound effect on other members of the littoral and shore communities. A large stand of Peltandra virginica just south of the outlet of the lake was mostly decimated when its former habitat dried as a result of the receding water.

The terrestrial plant which made the most rapid and striking gains in the newly exposed lake shore habitat was Lythrum salicaria. This species was present before the lake level was lowered in isolated patches, but has expanded to become by far the most abundant plant in the unshaded shore line. It now forms dense stands bordering the lake, making access to the lake difficult in areas which formerly had an unrestricted view.

In shaded parts of the newly exposed shoreline, Impatiens pallida is the most dominant invader. This plant was already an important component of nearby shaded habitats and merely expanded into the newly exposed areas.

Other species that were reduced in dominance by the lowering of the lake level were Sparganium eurycarpum, Scirpus validis, Vallisneria americana and Spagittaria rigida (Table 9).



KEY

P. crispus
Heteranthera
Ceratophylum
Anacharis
Najas
Vallisneria

1979 Biomass g ODW / m^2

1	2	3	4	5	6	7	8
–	–	–	–	–	–	0.9	2
–	–	–	–	–	–	–	–
–	0.6	1.5	–	–	–	–	–
–	–	–	–	–	–	–	0.4
–	280	–	–	–	–	–	340
–	–	–	–	–	–	–	–

Figure 20. Positions of vegetation zones along Southeast Transect before and after lake lowering. Note the 0.5 m difference in water depth. Najas became extremely dominant in the zone left by the receding P. crispus.

TABLE 9. COMMON VASCULAR PLANTS OF COLLINS LAKE ACCORDING TO VEGETATION ZONES (Relative abundance is listed before and after the restoration; R = rare, L = low, M = moderate, A = abundant).

Zone	Species	Abundance		
		Before	After	Net Change
Aquatic				
	Anacharis canadensis Michx.	L	L	0
	Ceratophyllum demersum L.	M	M	0
	Najas minor All	R	A	++
	Potamogeton crispus L.	A	A[1]	-
	Trapa natans L.	L	M	+
	Vallisneria americana Michx.	M	L	-
Emmergent				
	Lemna minor L.	M	M	0
	Nymphaea tuberosa Paine	M	L	-
	Peltandra virginica (Raf.) Blake	M	L	-
	Potamogeton natans L.	L	NONE	-
	Sagittaria latifolia Will	L	R	-
Moist Terrestrial				
	Erigeron annuus (L.) Pers.	L	M	+
	Hypericum punctatum Law.	L	M	+
	Impatiens pallida Nutt.	L	M	+
	Lythrum salicaria L.	L	A	++
	Sparganium eurycarpum Englem.	L	R	-
	Typha latifolia L.	L	M	+

[1] Low density in dredged area.

PHYTOPLANKTON

Algal biomass as estimated by chlorophyll a concentrations (Figures 21, 22) was low in the winter, increased during the spring bloom, decreased in May when P. crispus was growing and then increased during the summer months when nutrient levels and temperatures were high. This sequence was shown in all the stations and was repeated each year. Summer phytoplankton chlorophyll concentrations did not increase during the course of this project and may have even decreased slightly. Winter values appear to be slightly greater in 1978-79 than in previous winters.

Secchi disc depths (Figure 23) can reflect algal density, and annual changes closely follow those of chlorophyll a. Periods of greatest water clarity are the winter and the late spring when chlorophyll a levels were lowest. Winter water clarity was much greater in 1976-77 than in the following winters. This observation is supported by both the chlorophyll a data

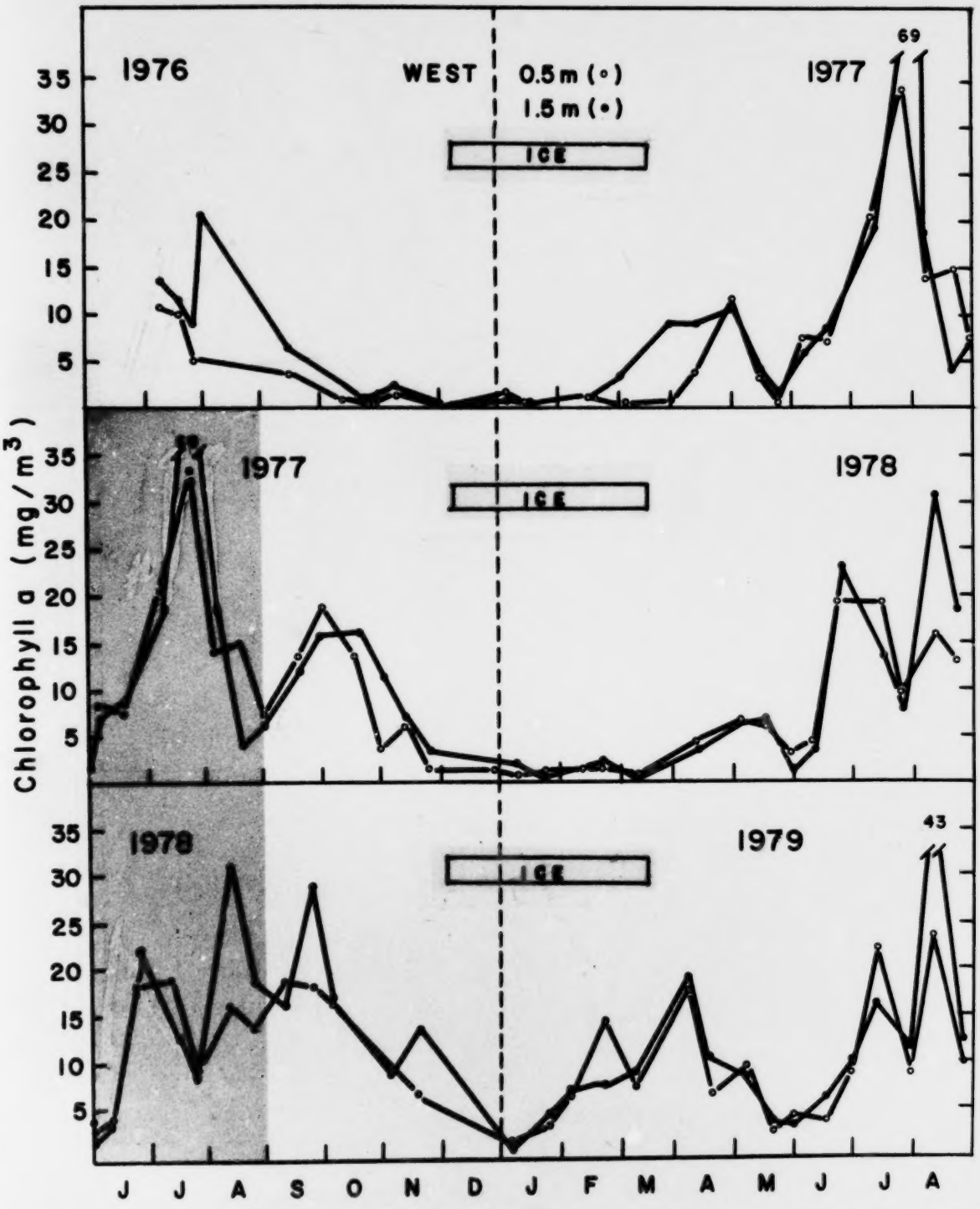


Figure 21. Chlorophyll a (mg/m^3) at 0.5 and 1.5 m versus time at West Station.

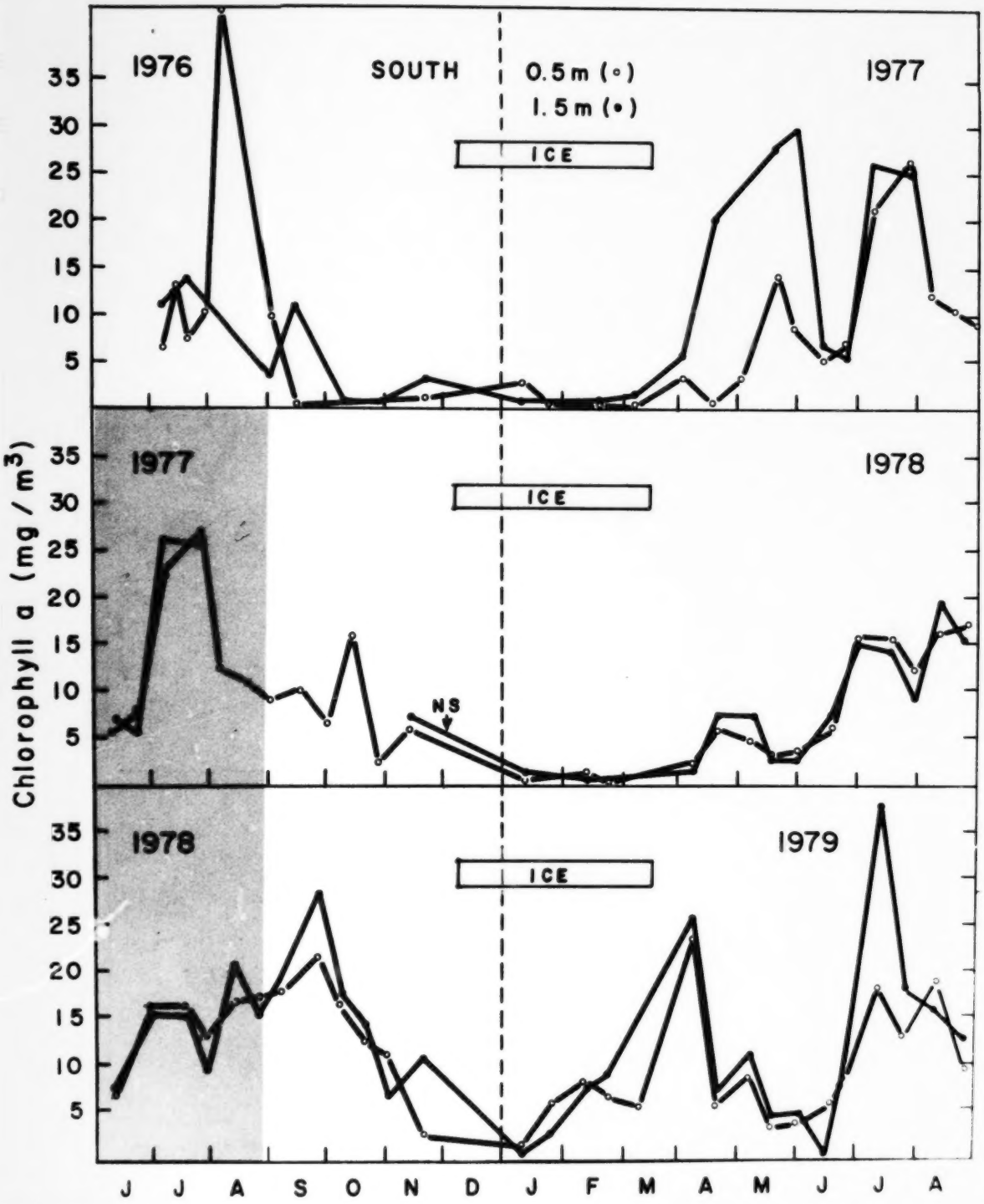


Figure 22. Chlorophyll a (mg/m^3) 0.5 and 1.5 m versus time at South Station. In November, 1977, this station was moved into the dredged region. Data for summer 1977 at 1.5 m are missing.

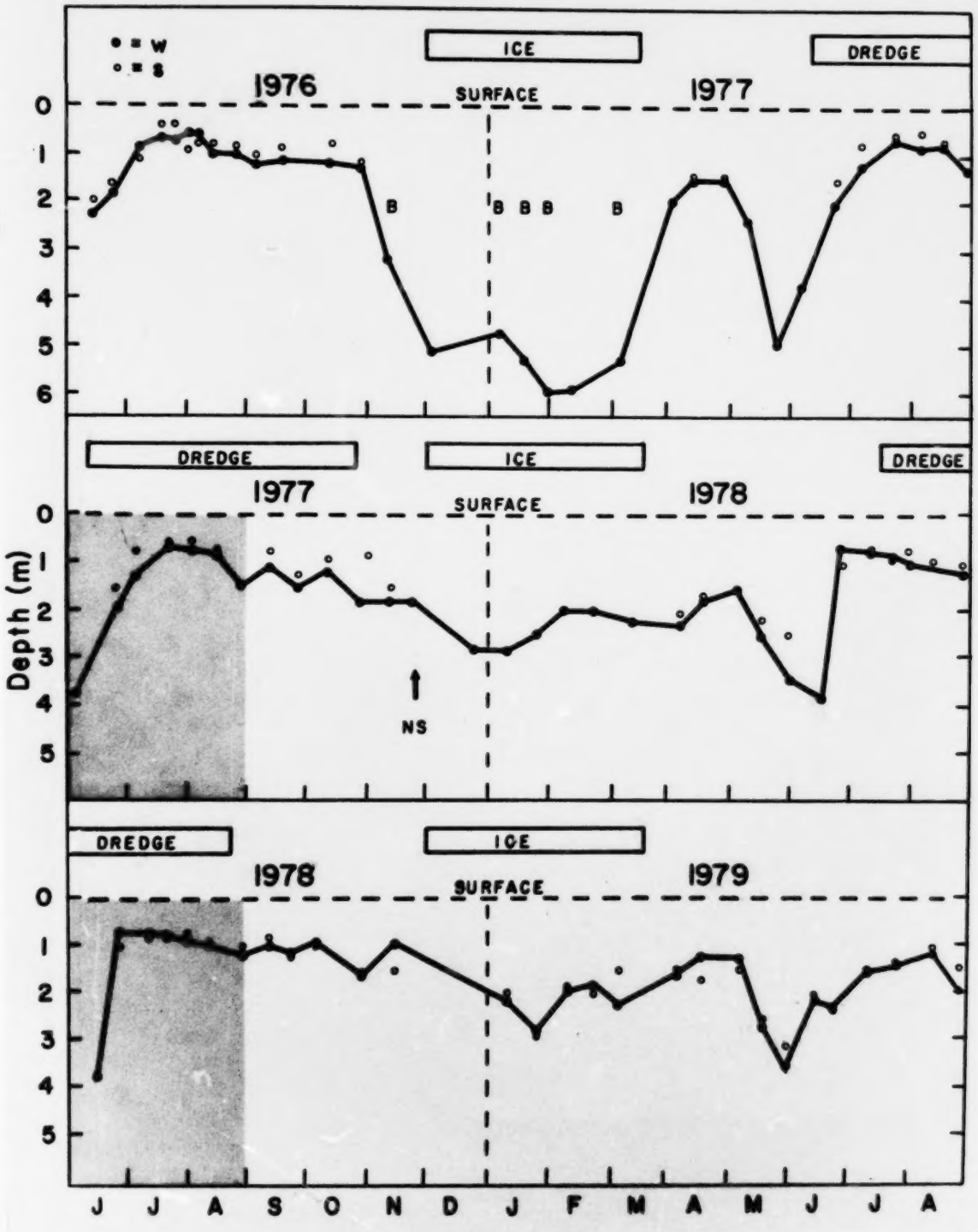


Figure 23. Secchi disc depths (m) versus time for West (dots) and South (circles) Stations. "B" indicates that the bottom was visible. South Station was moved at "NS".

(Figures 21, 22) and the cell counts found in Appendix C. Water clarity in the summer of 1979 was greater than at any other summer in the study.

Phytoplankton species found Collins Lake are listed in Appendix B. Species composition and cell counts for South Station (Appendix C) were typical of the other stations.

Pediastrum simplex is by far the dominant alga in Collins Lake, reaching populations of 10^7 cells/l and rarely decreasing to levels less than 10^4 cells/l (Figure 24). Its populations have been very consistent throughout the study period in both density and the time of bloom.

Chlorophyta totally dominated the summer months of this study (Figures 25, 26). Summer species composition was quite stable, with species of Pediastrum, Sphaerocystis, Oocystis, Scenedesumus, and Crucigenia consistently among the dominants. Notable by their absence in the summer were the Cyanophyta.

In the winter, divisions other than the Chlorophyta dominated, and the percent composition of the flora was not as consistent as it was in the summer. If there is a trend found in the three sampled winters, it shows a decrease of the Chryosphyta (mostly diatoms) and an increase in the Pyrrophyta (mainly Cryptomonads) and the Cyanophyta (mainly Chroococcus). The reason for this shift is not known.

In summary, the entire phytoplankton community has been fairly stable during the study period with no definite response to dredging. Algal biomass has not increased markedly, and the summer flora has been remarkably consistent.

ZOOPLANKTON

The zooplankton found in the lake are listed in Appendix D, and densities of the dominant species are listed in Appendix C for South 0.5 m. The species listed are quite common in many eutrophic lakes. The faunas of the South and West Stations were very similar.

Percent rotifers, cladocerans, and copepods on each sample date is shown in Figures 27, 28 for West 1.5 m and South 0.5 m samples. These two stations were similar and their annual cycles repeated each year of the study. Rotifers dominated during the summer and fall, whereas copepods dominated each winter and spring.

The copepod populations are characterized by the regular seasonal replacement of Mesocyclops edax in the summer with Cyclops bicuspidatus thomasi in the winter, and a fairly consant presence of Diaptomus birgei. The cladocerans were dominated by speices of Daphnia and Eubosmina. The rotifers were dominated by Keratella cochlearis during most of the year. Their number increased in regular pulses during the spring and summer.

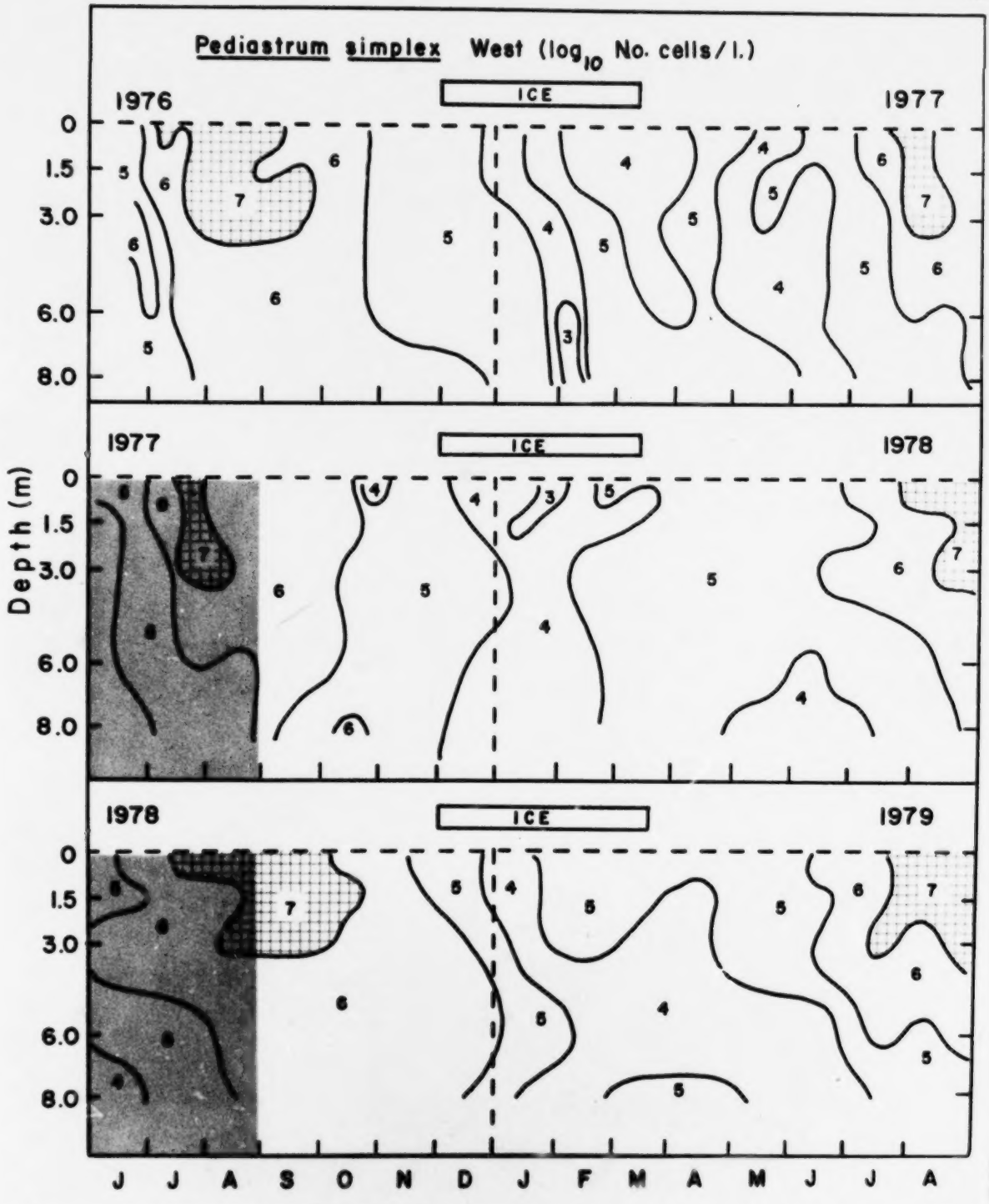


Figure 24. Pediastrum simplex density isopleths (log_{10} cells/l) at West Station.

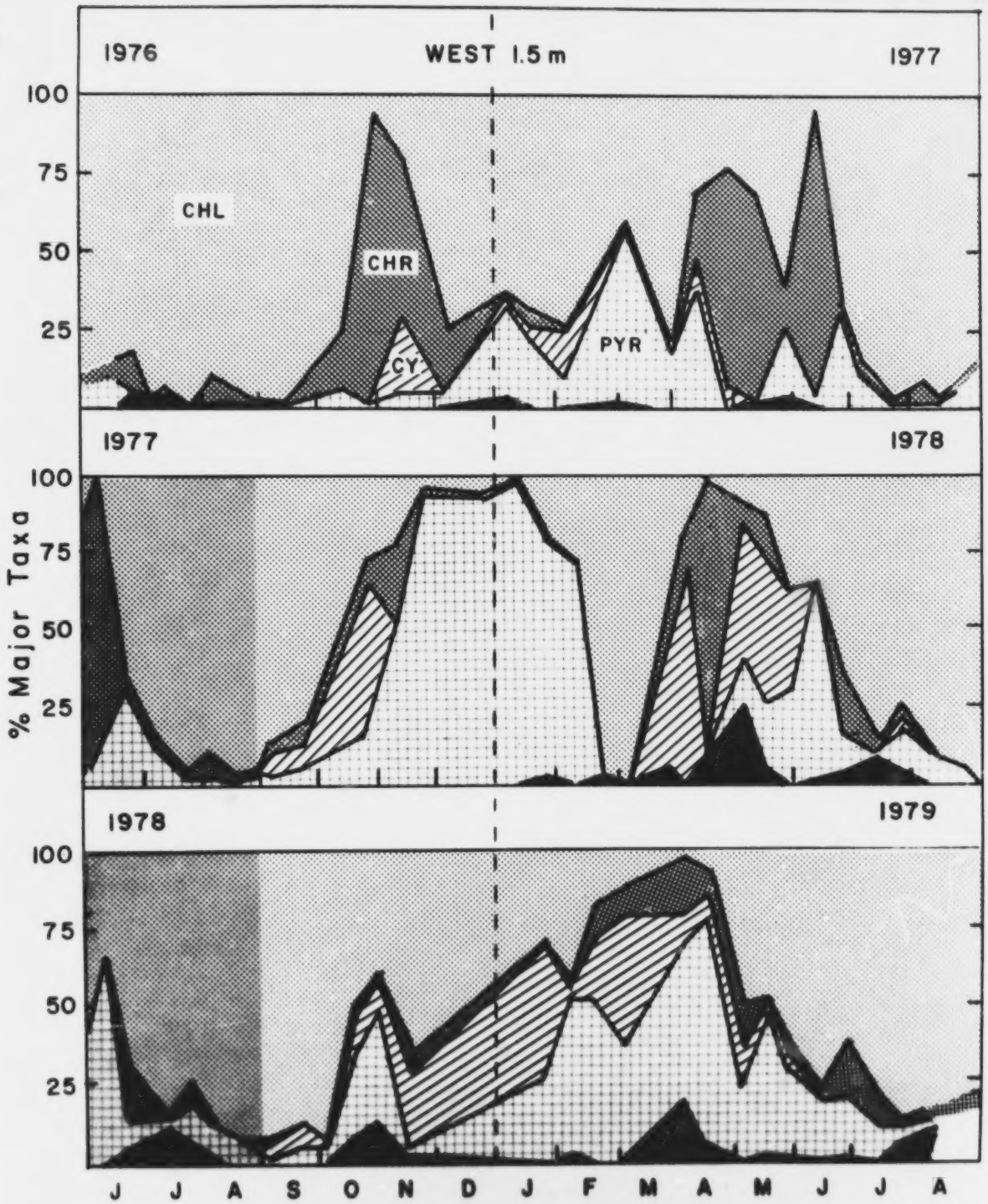


Figure 25. Percent major taxa of phytoplankton at 1.5 m at West Station. CHL=Chlorophyta, CHR=Chrysophyta, CY=Cyanophyta, PYR=Pyrrophyta, Black=Euglenophyta.

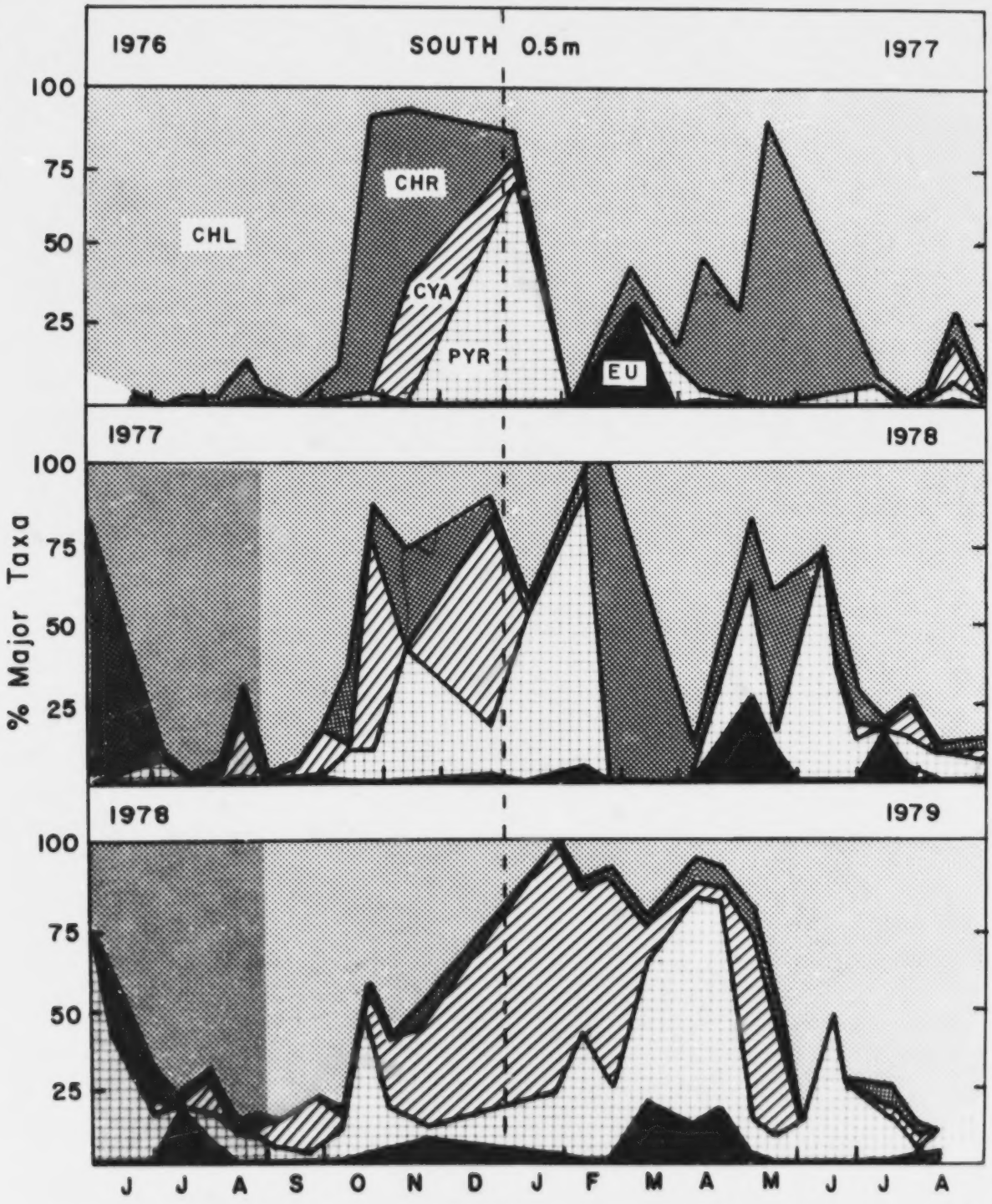


Figure 26. Percent major taxa of phytoplankton at 0.5 m at South Station. Abbreviations are the same as for Figure 25.

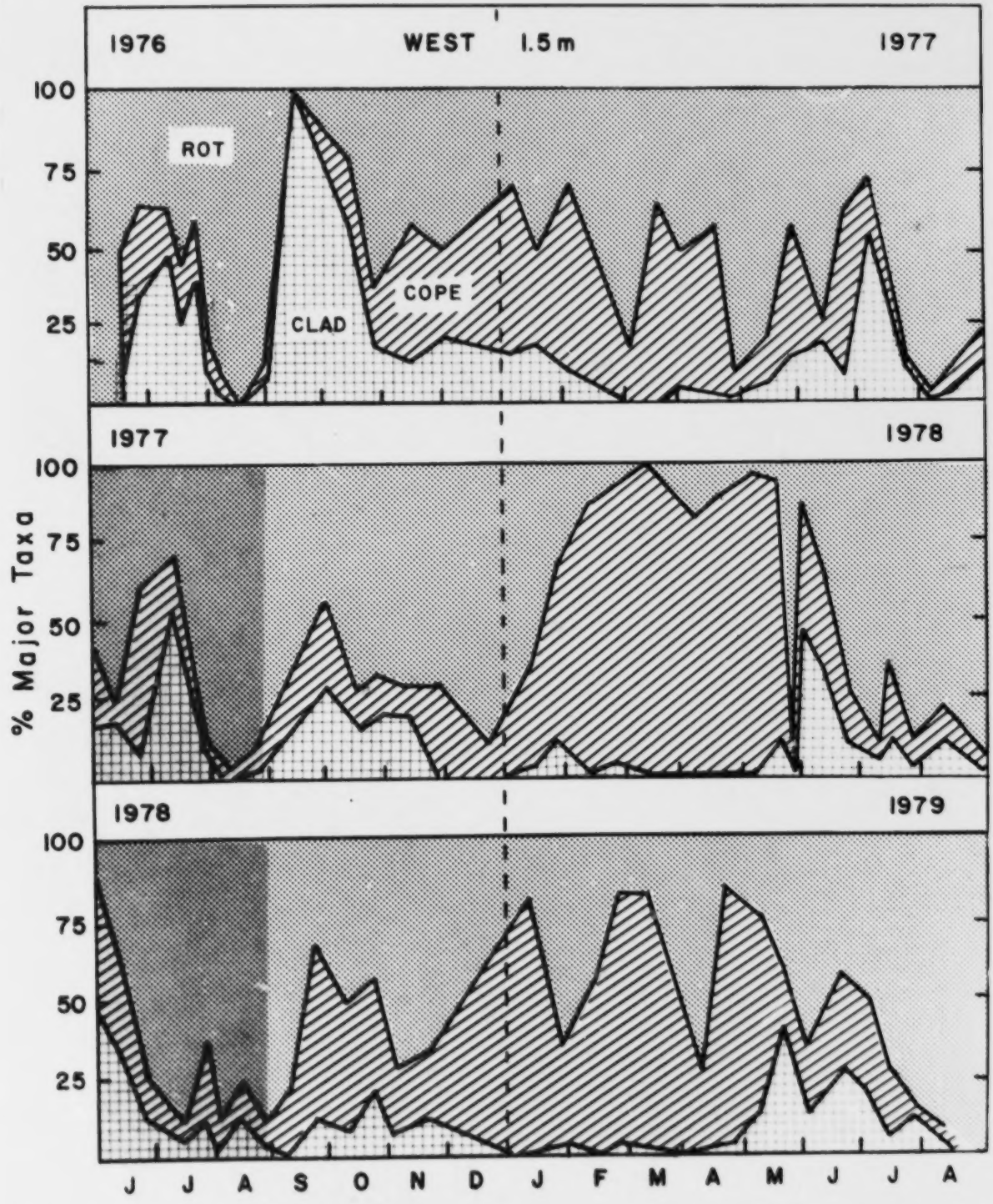


Figure 27. Percent rotifers (ROT), cladocerans (CLAD) and copepods (COPE) at 1.5 m West Station.

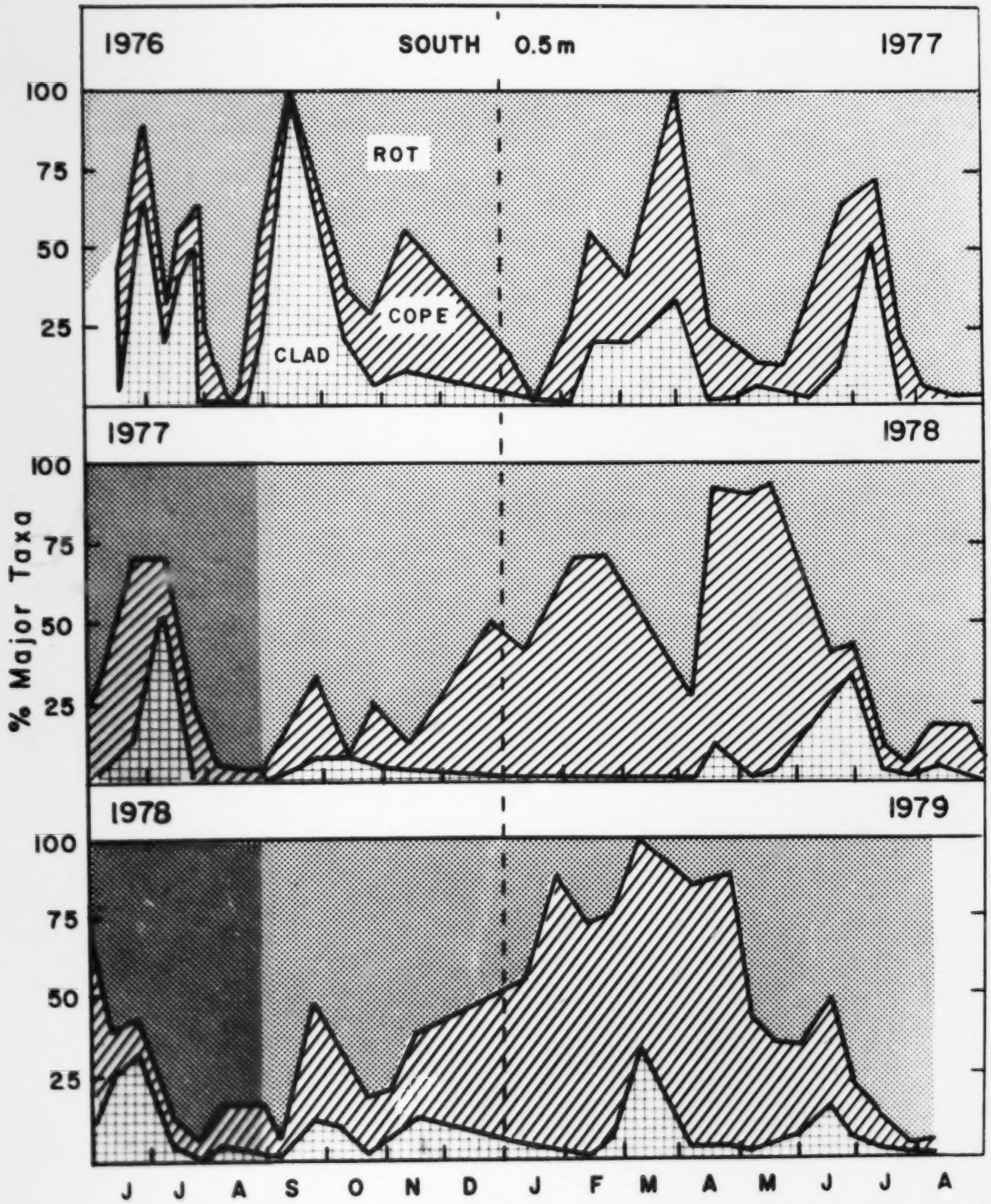


Figure 28. Percent rotifers (ROT), cladocerans (CLAD) and copepods (COPE) at 0.5 m South Station.

The Shannon-Weaver diversity index for the zooplankton community ranged from 3.5 to about 2.0 (Figure 29). Values fluctuated on a regular seasonal basis, but there was no systematic change during the study.

Since all regular samples were collected within a few hours of noon there was the concern that some mobile zooplankton with strong diurnal migration patterns might be missed. Intensive diurnal sampling at the West Station showed that no new taxa of zooplankton appeared that were not found in the noon sample.

In summary, the zooplankton community has been very stable during the study period. No new species appeared nor were there any changes in the percent of each group present in a given seasonal sample or in the diversity of the zooplankton community as a whole.

FISHES

Six hundred and twenty-eight fish were examined, most of which were yellow perch (Table 10). Catch per unit effort (CPUE) for both male and female yellow perch declined significantly for the "after" period relative to both the "before" and "during" periods. Decline in the number of male fish was especially notable. Significant changes in CPUE for the golden shiner were not detected.

TABLE 10. NUMBER OF MALE AND FEMALE YELLOW PERCH AND GOLDEN SHINER CAUGHT PER NET PER DAY (total number caught shown in parentheses).

	Yellow Perch		Golder Shiner	
	♂	♀	♂	♀
Before (8/6/76-7/6/77)	2.21 (135)	2.16 (132)	0.56 (34)	0.77 (47)
During (7/17/77-8/24/78)	1.61 (74)	3.87 (182)	0.72 (34)	0.98 (46)
After (9/7/78-8/31/79)	0.87 (45)	1.15 (60)	0.52 (27)	0.75 (35)

Close inspection by shore seine, night light spotting and diving revealed numerous golden shiner young but very few young perch. Gonadosomal indices (Figure 30), however, revealed typical maturation and spawning, both in timing and weights. Gonadosomal indices for golden shiner were also judged typical, but are not discussed here due to a limited sample size.

Indices of condition, both inclusive and exclusive of gonad weights, for the yellow perch (Table 11) showed a significant increase during dredging, but little or no change between "after" and "before" periods. Generally, conven-

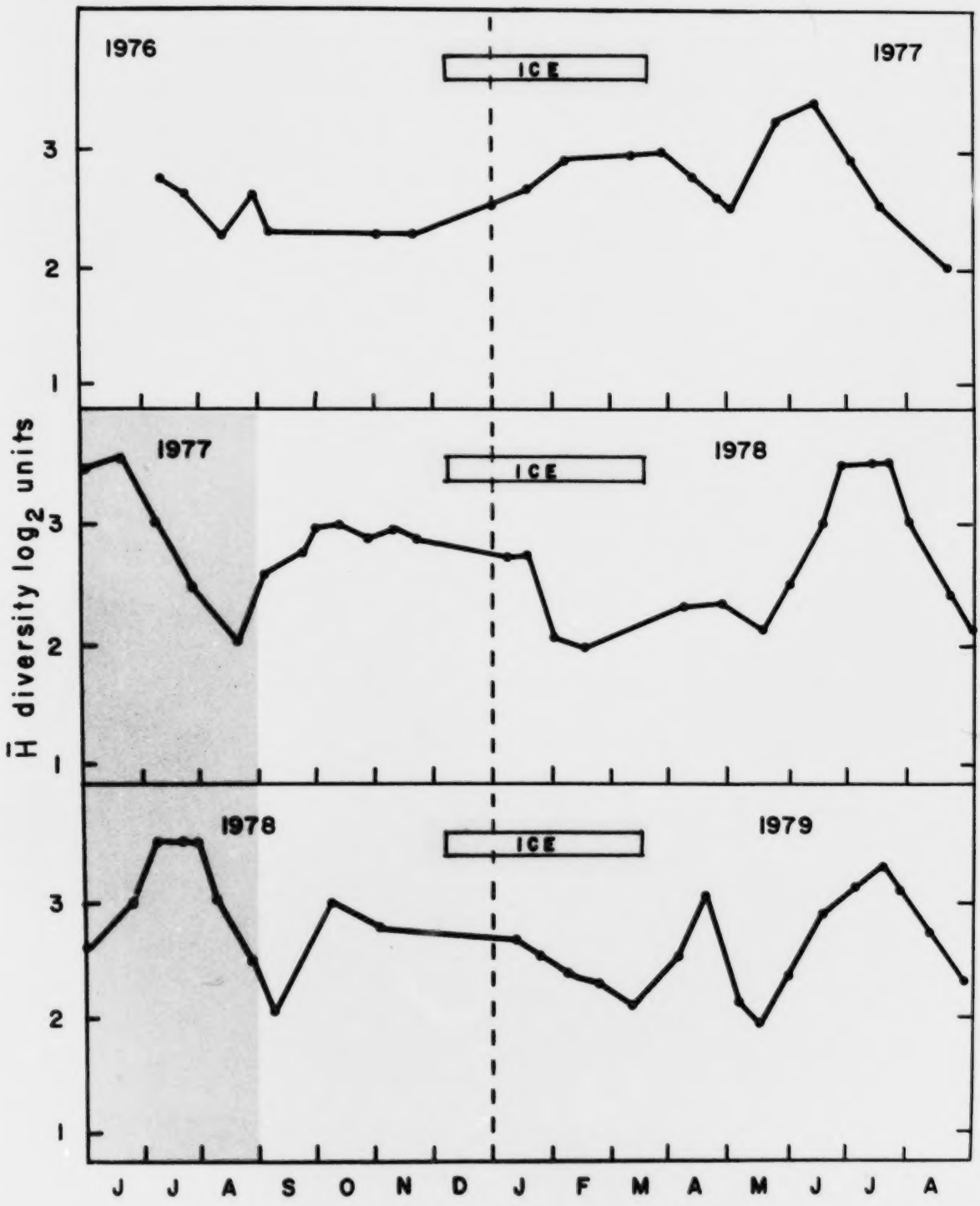


Figure 29. Zooplankton species diversity ($\log_2$ units) versus time for West Station.

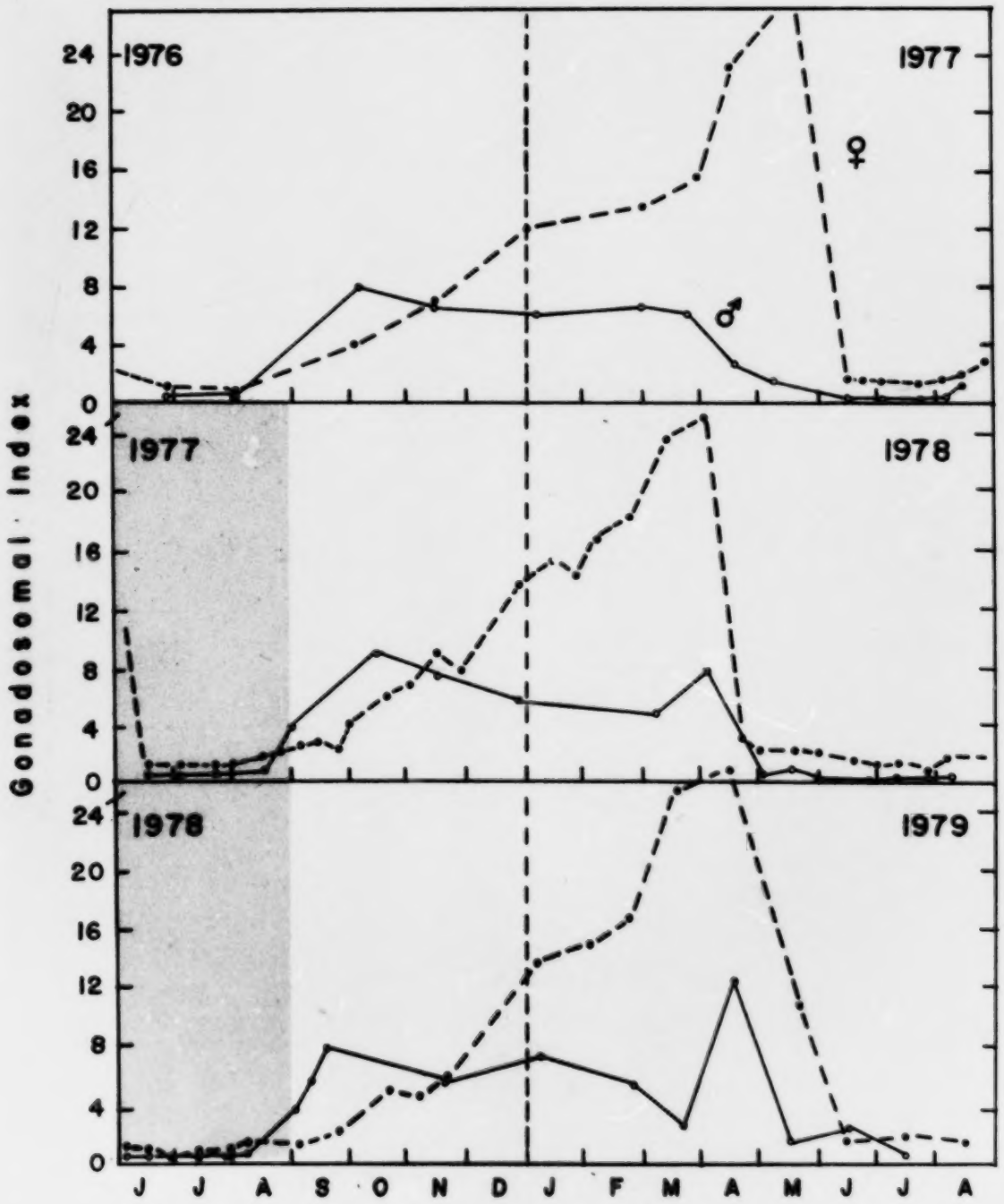


Figure 30. Gonadosomal index for yellow perch versus time.

TABLE 11. THE MORPHOLOGY OF YELLOW PERCH AND GOLDEN SHINERS (sample sizes in parentheses).

	Sampling Period			Significance of the Difference[a]		
	Before[b]	During[b]	After[b]	Before-During	During-After	Before-After
Index of Condition						
Yellow Perch	1.06 (208)	1.16 (246)	1.09 (68)	++	-	-
Golden Shiner	1.19 (82)	1.16 (74)	1.19 (63)	-	-	-
Index of Condition (gonad corrected)						
Yellow Perch	1.00 (148)	1.08 (174)	1.00 (66)	++	++	-
Golden Shiner	1.15 (44)	1.12 (69)	1.14 (61)	-	-	-
Average Age (in days)						
Yellow Perch	1381 (211)	1396 (248)	1501 (68)	-	-	-
Golden Shiner	1165 (82)	1331 (75)	1417 (63)	-	-	-
Hepatosomal Index						
Yellow Perch	1.77 (147)	1.34 (172)	1.47 (66)	+	-	-
Golden Shiner	2.20 (44)	2.88 (69)	2.10 (61)	-	-	-

[a] Using the T test, 95% C.I., ++ highly significant, + marginally significant; - not significant.
[b] Before, 6 Aug 76 to July 77; During, 18 July 77 to 24 Aug 78; After 7 Sept 78 to 21 Aug 79.

tional indices, i.e., gonadal weight included, of Collins Lake fish were well below those of most other studied populations (Table 12). The condition index of the golden shiner (Table 11) showed no significant change during the study period.

Average age for both species (Table 11) appeared to increase, but these differences were not significant.

Hepatosomal indices for the yellow perch (Table 11) suggested a marginally significant decline during dredging relative to the "before" period. Hepatosomal indices for the golden shiner did not change.

Macrophage ceroid deposition, indicated by the density of macrophage centers, suggested that the yellow perch in Collins Lake had a greater value for both liver and spleen than perch from other lakes and rivers in the area (Table 13). Ceroid indices reflecting the volume of ceroid accumulated each year increased significantly over the study period for yellow perch and decreased for the golden shiner (Table 14).

TABLE 12. CONDITION INDICES FOR YELLOW PERCH (after Carlander 1950).

Locale	Condition Index
Massachusetts	1.42
Massachusetts	1.28
Deep L., Michigan	1.03
Saginaw Bay, Mighigan	1.27
Massachusetts	1.29
North L., Michigan	1.46
Southern Wisconsin	1.52
Saginaw Bay, Michigan	1.29
Green Bay, Wisconsin	1.26
Michigan	1.17
Collins Lake, N.Y.	
"before"	1.06
"during"	1.16
"after"	1.09

TABLE 13. MACROPHAGE CENTERS PER mm^2 FOR THE LIVER AND THE SPLEEN OF YELLOW PERCH IRRESPECTIVE OF AGE (sample size).

Locality	Liver ♀	Liver ♂	Spleen ♀	Spleen ♂
Collins Lake	1.66 (157)	2.26 (97)	15.62 (114)	16.09 (58)
Jenny Lake, Adirondacks	2.25 (8)	2.48 (2)	7.49 (8)	8.63 (2)
Stillwater, Hudson R.	1.10 (8)	--	3.89 (8)	--
Big Boom Bay, Hudson R.	0.95 (11)	0.90 (11)	2.27 (11)	5.32 (11)
Lake George, Adirondacks	0.09 (3)	--	8.71 (3)	--
Polliwog Pond, Adirondacks	0.05 (7)	--	1.90 (7)	1.48 (1)
Gilman Lake, Adirondacks	1.25 (2)	0.46 (1)	13.01 (2)	1.56 (1)
Schenectady, Mohawk R.	0.02 (2)	0.13 (1)	5.88 (2)	1.45 (1)
Lake Sunapee, NH (Sexes Combined)	0.3 (5)		3.38 (5)	

Collins Lake, Schenectady Co., N.Y.; Jenny Lake, Saratoga Co., N.Y.; Stillwater vicinity, Hudson R., Saratoga Co., N.Y.; Big Boom Bay, Glens Falls Vic., Hudson R., Warren Co., N.Y.; Lake George, Gull Bay area, north basin, Warren Co., N.Y.; Polliwog Pond, Upper Saranac Lake vicinity, Franklin Co., N.Y.; Gilman Lake, Lake Pleasant vicinity, Hamilton Co., N.Y.; Lake Sunapee, Sullivan Co.-Merrim Co., N.H.; Schenectady (sector) Mohawk R., Schenectady Co., N.Y.;

TABLE 14. THE CEROID INDICES FOR THE SPLEEN LIVER AND KIDNEY OF YELLOW PERCH AND GOLDEN SHINERS (sample size shown in parentheses).

Organ	Species	Sampling Period			Significance of the Difference[a]		
		Before[b]	During[b]	After[b]	Before-During	During-After	Before-After
Spleen	Yellow Perch	.88 (185)	1.48 (195)	1.86 (68)	++	+	++
	Golden Shiner	1.10 (68)	1.01 (67)	.94 (63)	-	-	-
Liver							
	Yellow Perch	.39 (168)	.45 (188)	.86 (65)	+	++	++
	Golden Shiner	.79 (64)	.37 (52)	.28 (48)	++	++	-
Kidney							
	Yellow Perch	.43 (180)	.47 (193)	.94 (67)	-	++	++
	Golden Shiner	.98 (69)	.47 (62)	.50 (55)	++	-	++

[a] Using the T Test, 95% C.I., ++ highly significant, + marginally significant, - not significant.
[b] Dates as in Table 11.

DISCUSSION

The intent of this study was to determine the effects of hydraulic dredging on Collins Lake by comparing data gathered before, during and after the dredging. These comparisons are clouded, however, by many factors beyond the control of the research team. The brevity of the study period allowed only one "before" and one "after" growing period to be studied. We were therefore unable to determine the magnitude and timing of natural fluctuations in the lake to use as a context in which to interpret observed changes during the study. The lowering of the lake level most certainly altered zonation in vascular plant communities, and other more subtle effects were possible. Two "one hundred-year" floods occurred during the study, backing considerable amounts of Mohawk River water into the lake each time. Perhaps most important, much less of the lake bottom was dredged than was originally projected. Dredging effects that might appear in a more extensively dredged lake might have been covered by natural variation in the studied parameters. The following discussion must be read with these limitations in mind.

Changes attributed to dredging will be discussed regarding the physical, chemical and biological aspects of the lake. Since the main reason for dredging the lake was to remove nutrient rich sediments, reduce the growth of P. crispus and control other nutrient inputs to the lake, emphasis was placed on developing a phosphorus budget for the lake, the nutrient most likely limiting algal growth.

PHYSICAL CHANGES

Approximately 55,000 m^3 of sediments were removed from the lake, deepening about 2.6 ha (10% of the lake area) to about 3 m (Figure 4). Early into the dredging village officials decided to dredge to the maximum depth capability of the dredge, rather than maximizing areal extent by more shallow dredging. The latter strategy would have removed more of the highly organic surficial sediment material. This decision was based on: (1) a desire to ensure the control of P. crispus in the dredged area; (2) the labor required to move the dredge guide wires and resultant reduction in dredge time efficiency; (3) the large number of stumps remaining in the shallower area which would have further reduced dredging efficiency. The deeper sediments were more dense than surficial sediments, so the sedimentation basin filled more rapidly than anticipated, forcing the termination of dredging in August, 1978.

Dredging to the north of the island was stopped when the dredge struck a concentration of submerged Indian artifacts, mostly net sinkers. Upon consultation with the New York State archeologist, an archeological "preserve" was established, and dredging was resumed about 15 m to the east, resulting in two separate dredged basins to the north of the island.

The most conspicuous change in the watershed is the presence of the sedimentation basin. During the project there were no objectionable odors, so the proximity of the basin to residential areas created no problem. The basin could be cleaned out and re-used in the case of future dredging.

CHEMICAL CHANGES

Broad variations in the studied chemical parameters were rarely conspicuous and often best explained in terms of unusual sampling events. For example, during winter (through the ice) sampling, the inadvertent disturbance of the bottom in the course of plankton sampling probably resulted in several exceptionally large peaks of total phosphorus. Further, the lowering of lake level resulted in the northern station being especially shallow and thus vulnerable to bottom disturbance.

Dissolved oxygen and percent saturation of dissolved oxygen appeared to vary between the undredged North Station to the dredged South Station (Figure 8). In 1976 both stations were undredged and showed extreme undersaturated values in June corresponding to the decomposition of the dead P. crispus. In 1977 and 1978, the undredged North Station showed a greater depletion (60 percent saturation) of oxygen during this decomposition period and in 1979, a drastic depletion (25 percent saturation) was noted. During the earlier time periods (1977-1978), the dredged South Station was below 90 percent saturation while in 1979, it was always 100 percent or greater. The absence of decaying P. crispus from the dredged South Station appeared to help oxygen demand during the June decaying period. In the future, South Station should be able to maintain near saturated conditions.

During the summer, dissolved oxygen near the bottom (3 meter depth) of the dredged South Station was depleted more than at North Station. The degree of undersaturation was less in 1979 than in 1978 near the bottom, which suggests that uptake of oxygen was decreasing as the newly exposed bottom was coming to an equilibrium condition.

Concentrations of total phosphorus and orthophosphorus showed a marked variation between the North and South Stations (Figures 14, 16). Prior to dredging, both areas were reasonably similar in their seasonal variation. After South Station had been dredged, the tremendous increases in both ortho and total-P, which continued to occur at North Station at the time of the decomposition of P. crispus, were no longer evident at South. The dredged area therefore yields less phosphorus to the lake water for subsequent algal growth.

Nitrate concentrations show marked differences in two periods of the year (Figure 12). Under the ice, concentrations of NO_3-N were lower at the dredged South Station than at North Station. This is attributed to higher amounts of organic nitrogen in the sediment of the undredged area.

BIOLOGICAL CHANGES

Both the phytoplankton and zooplankton communities were remarkable in their stability during the study period. There was no increase in chlorophyll *a* concentrations, no change in the dominant zooplankton or summer phytoplankton and no decline in zooplankton diversity. The one change was the increase in the Cyanophyta during the winter (Figures 25, 26), almost entirely from increasing populations of *Chroococcus*. Because of its small size the enumerative methods used in this study overestimate its ecological importance to the lake. On a biomass basis, this alga would be relatively much less important.

Certainly the major change in the macrophyte populations of the lake was the 99% reduction of *P. crispus* in the 2.6 ha of dredged bottom (Table 8). Apparently, the deepening of the lake to about 3 m, perhaps through reduction of light or temperature and alteration of substrate, suppressed both its persistence in and reinvasion of the dredged sector. At the same time, the population in the undredged area grew more productively. The basis for this increased growth in the undredged areas was obscured by the fall in lake level. The shifts in vascular plant zonation and expansion of *Lythrum salicaria* along parts of the shore were the direct result of lowering the lake rather than the dredging.

Characteristics of the populations of yellow perch, a carnivore and the golden shiner, an herbivore zooplanktivore, changed to some extent during the study. Catch-per-unit-effort declined for the male yellow perch (Table 10); probably reflecting a decline in actual numbers and failure of recruitment for the population. This seems best explained in terms of a failure of either eggs or fry to survive, because gonadal development and time of spawning were apparently typical for the species. This decline may not be ascribed to the dredging project, however, because it could have been underway at the onset of the study.

Conventional indices of condition revealed little change other than a slight increase for perch. Hepatosomal indices, however, declined between the "before" and "during" periods (Table 11) suggesting reduced nutrient availability (Larsen 1973; Voltonen 1974; Heidinger and Crawford 1977). The golden shiner, in contrast, remained relatively stable in terms of conventional indicators.

More interesting, ceroid indices (Table 14) showed a broad trend of ceriod increase for the key organs of spleen, kidney, and liver for the yellow perch, suggesting an increase in moribund cells and cellular debris (Roberts 1975; Ellis *et al.* 1976). In contrast, the golden shiner showed a declining ceroid index suggesting reduced trauma. It is tempting to suggest that lake conditions were shifting from those favoring the yellow perch to those favoring the golden shiner, but the role of dredging cannot be specified.

The physiological mechanisms involved in the observed changes of ceriod were not defined, but the potential for using the ceriod loading of key organs as an indication of trauma, here attempted in a routine way for wild fish populations for the first time, may merit further attention.

PHOSPHORUS BUDGET

The hydraulic dredging of Collins Lake was directed toward (1) the recovery of essential lake volume and (2) removal of sediments bearing nutrients available to both P. crispus and the phytoplankton. To assess the merits of the actual work done in the second regard, the phosphorus budget was developed for the lake.

Components in the phosphorus budget of Collins Lake (Figure 31) were calculated using a simple yearly averaged mass balance. The mass input (in kg/year) for each measured parameter was calculated or approximated and was compared to the total observed mass output. The difference was attributed to release of phosphorus from the sediment.

Atmospheric input was determined using the monthly rainfall volume onto the lake (measured at the Albany County Airport - 10 miles away) multiplied by the monthly concentration of phosphorus in the rainfall (data also from the Airport, determined by the USGS). The values for each water year (October to September) were - 1974-75; 4.3 kg/yr, 1975-76: 9.0 kg/yr, 1976-77: 12.4 kg/yr, and 1977-78: 4.4 kg/yr. The average (aritmetic) input of phosphorus was 7.5 kg/yr.

Groundwater, from the 40 or 50 springs, is the only sustained flow into the lake. The flow (Table 3) for each month was multiplied by the concentration of total phosphorus in the groundwater. Values for 1977-78 and 1978-79 were 25.1 and 42.5 kg/year, respectively. This gives an average input of 33.8 kg/yr of phosphorus to the lake from the groundwater.

Sporadic flooding of the lake from the Mohawk River occurred even though a tide gate (flapper valve) was in place at the outlet. An estimate of flood input flow used a measured flow value of 57 to 85 l/sec occurring during flooding. From past Mohawk River flooding data, an average of 5 days/year was used when substantial inflow would occur. This yielded an inflow of 3.7×10^7 l/yr and resulted in a lake rise of 15 cm during the flooding. Additional rise in lake levels observed during river inflow was attributed to groundwater, rain, and stormwater inflow to the lake. The combined effect of all these inputs would cause the lake level to rise about 30 cm (on the average) during each yearly flood. Using this flow input and the average concentration of 100 µg/l of phosphorus in the river water during the spring (USGS 1978) one obtains an average yearly river input of 3.6 kg/year.

The phosphorus input to the lake from the stormwater pipe at the east end of the lake only occurs during flooding when the tide gate (flapper valve) is closed or river water is flowing into the lake. Normally, the stormwater flows 10 meters directly to the outlet and does not circulate into the lake. Using an average value of phosphorus in stormwater of 1000 µg/l, stormwater flow estimated at 2.4×10^5 liters/day (using the area of the stormwater drainage basin and average rainfall-runoff values for a typical residential area) and 2.5 days (half the period during which river flooding occurred) of stormwater flow into the lake, the yearly average input of phosphorus was 0.6 kg/year.

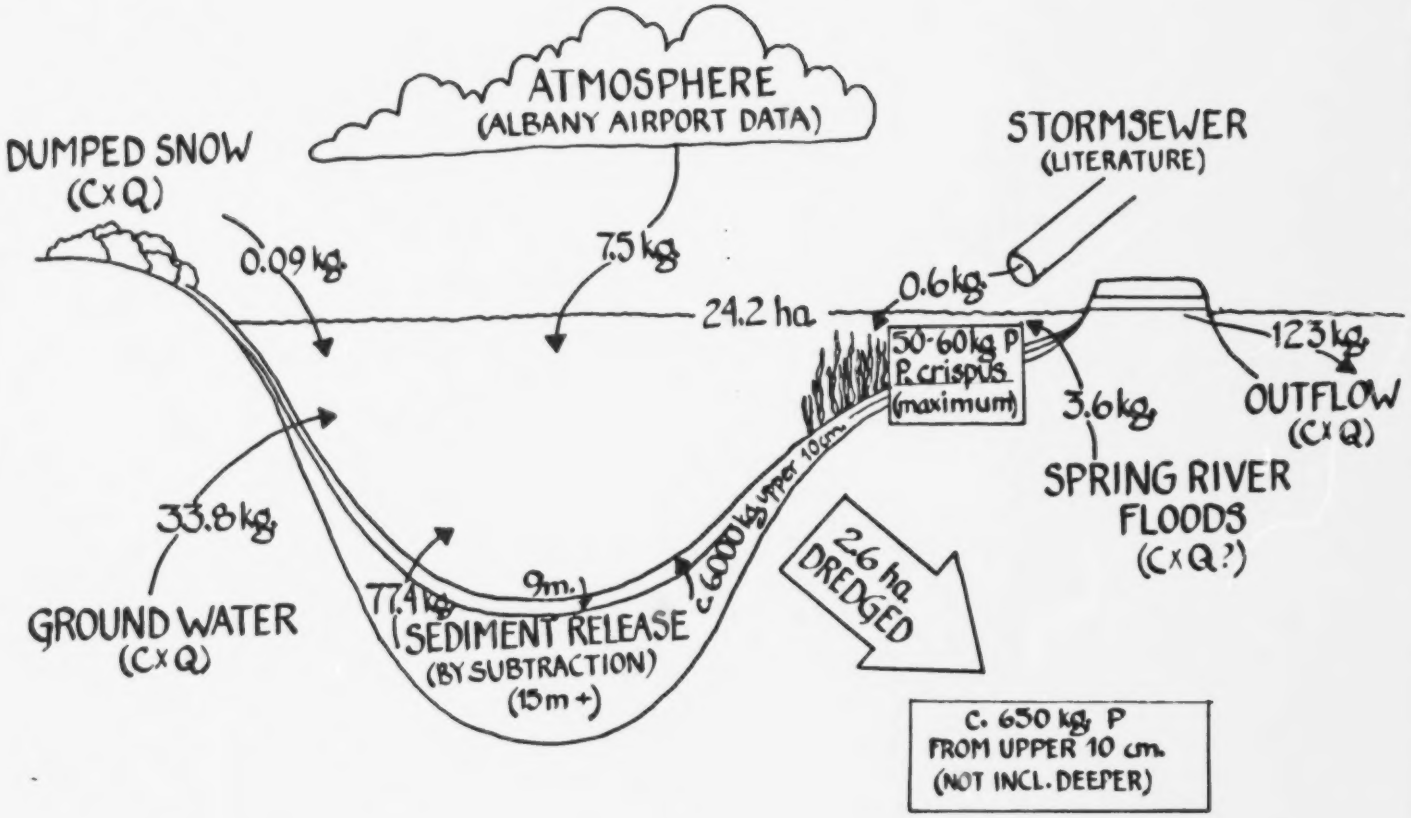


Figure 31. Mass balance diagram for phosphorus in Collins Lake.

Snow dumped into the parking lot contained phosphorus which flowed into the lake when the snow melted. Values of phosphorus determined in the snow were averaged and multiplied by the water content of the dumped snow. Assuming all of this reached the lake, the input of phosphorus would be 0.09 kg/year.

The total of all these inputs (Table 15) is about 45.6 kg of phosphorus entering the lake every year. The output, the tide gate, was measured for flow on a monthly basis (Table 3). These flow values were multiplied by the average monthly concentrations of total phosphorus for the outlet (East) Station. This gave an average output of 123 kg of phosphorus each year. The difference between total input (45.6 kg/yr) and output (123 kg/yr) can only be attributed to the net release of 77.4 kg phosphorus/yr from the sediment. The actual release of phosphorus from the sediment back into the water is probably much greater than 77.4 kg/year because deposition is also occurring.

Phosphorus flux from the sediment was not measured directly in Collins Lake. By using average net release rates obtained from other studies (summarized in Snow 1976) we can estimate what this annual flux might be. Eighty percent of Collins Lake has aerobic sediments all year. Using 0.5 mg total P/m^2 day as an average net release rate for a typical mesotrophic lake, we calculated 35.5 kg P would be release each year from this part of the lake. The other 20% of the lake is aerobic for eight months (5.8 kg/yr released) and anaerobic for the other four months (5 mg total P/m^2 day for a total of 29.2 kg/yr). The total release from the lake calculated in this way would be 70.5 kg/yr which agrees closely with the 77.4 kg/yr calculated previously.

A third method to determine the net release from the sediment is a mass balance lake model incorporating release and sedimentation of phosphorus (Table 16; Snow and DiGiano 1976; DiGiano and Snow 1977).

Using values generated for the undredged lake and from the literature (Table 17), and an average interstitial water phosphorus concentration of 250 mg/m^3, one obtains a value of 31.5 µg/l (eq. 2) for the lake water concentration. This is reasonably close to the actual measured average (excluding flooding effects) of 27.6 µg/l. Using the fourth term in Equation 1, $eAK_1(P_i-P_l)$, which describes the theoretical sediment flux or release of P into the lake water, and P_l=31.5 µg/l, one obtains a value for this flux of 3.86×10^6 mg/day or 15.9 mg/m^2-day. Using the third term in Equation 1, K_2P_lV, which describes the overall sedimentation of P into the sediment, one obtains a rate of 2.65×10^6 mg/day or 15.02 mg/m^2-day. The net release of phosphorus from the sediment would then be 0.88 mg/m^2-day, giving a total phosphorus flux from the lake sediments of 77.2 kg/yr.

The near agreement of the derived values of sediment release with that actually measured in the lake suggests that the release rates used in the first calculation and K_1 and K_2 used in the second calculation were close to those found in Collins Lake. All three methods thus indicate that about 63 percent of the phosphorus in the lake water (and output) is derived from the sediment and only 27 percent comes from the inflow (groundwater).

TABLE 15. PO_4 BUDGET FOR COLLINS LAKE (see the text and Tables 16 and 17 for derivations).

I. Average Mass into Lake

1) Atmosphere - 7.5 kg/yr

2) Groundwater - 33.8 kg/yr

3) Flood water - 3.5 kg/yr

4) Stormwater - .6 kg/yr

5) Dumped snow - .09 kg/yr

6) Total in = 45.59 kg/yr

II. Total outflow = 123 kg/yr

III. Sediment Release = 77.4 kg/yr

IV. Release approximated from sediment flux rates (Snow, 1976)

1) Anaerobic sediment release = 5 mgP/m^2-day

2) Aerobic flux rate = .5 mgP/m^2-day

3) Assuming 20% of lake (West) is anaerobic for 4 months/yr - 29.2 kg/yr

4) Assuming 80% of lake (shallow stations) are aerobic 12 months/yr - 3.55 kg/yr

5) Assuming 20% of lake (West) is aerobic for 8 months/yr - 5.8 kg/yr

Total sediment flux rate = 70.5 kg/yr

TABLE 16. MASS BALANCE EQUATION FOR PHOSPHORUS (after the method of Snow and DiGiano 1976).

$$QP_{in} - QP_{out} - K_2 P_l V + eAK_1 (P_i - P_l) = V \frac{dp}{dt} \qquad \text{(Equation 1)}$$

Integrated form: $t = \infty$, $P_{lake} = P_{out}$, assumes complete mixing

$$P_l \text{ (equilibrium)} = \frac{P_o + \frac{eA}{V} K_1 P_i \bar{T}}{1 + \frac{eA}{V} K_1 \bar{T} + K_2 \bar{T}} \qquad \text{(Equation 2)}$$

Q = Water flow (m^3/day)

P_o = P_{in} = concentration of P of inflow (µg/l), (mg/m^3)

P_l = P_{out} = concentration of P of outflow (µg/l), (mg/m^3)

K_2 = Phosphorus sedimentation constant, .176/day

P_l = Concentration of P in lake water (µg/l), (mg/m^3)

V = Volume of lake (m^3)

e = Porosity of sediment (dimensionless)

A = Area of lake surface (m^2)

K_1 = Phosphorus release constant, .091 m/day

P_i = Concentration of P of Interstitial Water (µg/l), (mg/m^3)

$\bar{T}$ = Dentention time, days

(P denotes concentration of total phosphorus as P)

conventional indices, i.e. gonadal weight included, of Collins Lake fish were well below those of most other studied populations (Table 12). The condition index of the golden shiner (Table 11) showed no significant change during the study period.

Average age for both species (Table 11) appeared to increase, but these differences were not significant.

Hepatosomal indices for the yellow perch (Table 11) suggested a marginally significant decline during dredging relative to the "before" period. Hepatosomal indices for the golden shiner did not change.

Macrophage ceroid deposition, indicated by the density of macrophage centers, suggested that the yellow perch in Collins Lake had a greater value for both liver and spleen than perch from other lakes and rivers in the area (Table 13). Ceroid indices reflecting the volume of ceroid accumulated each year increased significantly over the study period for yellow perch and decreased for the golden shiner (Table 14).

The opposite situation could be predicted if the entire lake bottom were dredged and the interstitial phosphorus of the newly exposed sediment was only 85 µg/l (Table 17). Lake water concentration would then only be 11.1 µg/l as calculated by the model. The lake sediment would then remove more than it released. This was the overall objective to dredging the lake although not all of the bottom could be dredged. If 65 percent of bottom had actually been dredged, thus shifting 65 percent of the bottom from an interstitial P level of 250 to that of 85 µg/l, the resulting lake water phosphorus concentration would drop from 31.5 µg/l to 18.2 µg/l as calculated by the model. Besides decreasing the total phosphorus in the lake water, the dredging would have removed the highly organic sediment and deepened the lake probably reducing the growth of macrophytes.

TABLE 17. CALCULATIONS FOR LAKE PHOSPHORUS USING MASS BALANCE EQUATION (developed by Snow 1976).

Variable	Value	Source
P_o	11.3 µg/l	This study
A^o	243,000 m^2	This study
$\bar{V}$	669,300 m^3	This study
$\bar{T}$ (flow-96 liter/sec)	84 days	This study
$*K_1$	.091 m/day	Snow 1976
$*K_2$	.176 /day	Snow 1976
e (dredged)	.775	This study
e (undredged)	.815	This study
P_i (dredged)	85.0 µg/l	This study
P_i (undredged)	250 µg/l	This study
P_1 (actual average)	37.6 µg/l	(average of all east data, except when inflow occurred)

* Rate constants K_1 and K_2 were determined in situ in a lake similar to Collins Lake.

Two different approaches were used to approximate the total reservoir of phosphorus in the sediment. Both approaches assume that the top 10 cm of sediment is the reactive zone for phosphorus transfer to the lake water (Snow and DiGiano 1976). For determinations of the total amount of soluble PO_4 in the interstital water of the sediment, an average concentration of 250 µg/l was used. In this zone, about 80 percent of the volume is water (i.e., porosity is 0.80). Using these concentration and water volume figures, one can determine an available interstitial phosphorus concentration of 2×10^4 $\mu gP/m^2$ or 4.8 kg of phosphorus available throughout the reactive lake bottom which is soluble and in the interstitial water and could ultimately diffuse into the lake water.

The determination of the total amount of phosphorus in this 10 cm layer used the value of 20 percent of the sediment volume as solids (i.e., 80 percent is water). An average concentration of 0.5 mg P/gm dry weight of sediment multiplied by the dry weight of sediment in a 1 m x 1 m x 10 cm reactive sediment yielded 2.5×10^4 $mg/P/m^2$ or about 6000 kg of total phosphorus incorporated in the top 10 cm of lake sediment. Since there is an exchange of phosphorus from the solid phase (6000 kg) to the interstitial water (4.8 kg) and from sediment interstitial water to the lake water, one of the main objectives of the project was to remove much of this highly exchangeable phosphorus sediment from the lake. Since the sediment is losing phosphorus at a rate of about 70 kg/yr, this reactive zone would eventually be depleted of much of its phosphorus. As the sediment phosphorus diminishes, however, the release rates would decrease, slowing the annual phosphorus loss of the sediments.

The dredging removed sediment from about 2.6 ha of the lake bottom or about 650 kg of sediment bound phosphorus. The newly exposed bottom sediments have much less interstitial phosphorus (85 µg/l versus 250 µg/l) which would indicate that much less exchangeable phosphorus is available in the newly exposed sediment. The newly exposed sediment had about the same amount of total phosphorus on the solid phase (0.45 mgP/gm oven dry weight) but much of this must be mineralized or bound in such a way as to be unexchangeable with the interstitial water.

Thus the overall effect of dredging was to remove bottom sediment, high in exchangeable phosphorus, thus lowering the concentration in the interstitial water. This results in a decreased flux into the lakewater. We did not find a marked decrease in total phosphorus levels as the result of dredging in Collins Lake, most likely because only 10% of the sediments were dredged and annual variabilities in the total phosphorus data masked the effect. Dredging only 10% of the sediments would result in only a 2 µg/l decrease in total phosphorus in the lakewater.

PHOSPHORUS AND PLANT GROWTH

Potamogeton crispus is a key compartment in the phosphorus system in Collins Lake. Its biomass in the spring has been estimated at about 20×10^3 kg ODW. If about 0.30% of that is phosphorus (Table 18), then 60 kg of phosphorus will be cycled through this compartment in the lake each year. This flux is not gradual, but occurs in pulses as P. crispus goes through its stages of rapid growth and decomposition.

TABLE 18. BIOMASS AND PHOSPHORUS CONTENT OF P. cripus (1977)

Date	Phosphorus Content of P. crispus (% ODW)	Biomass P. crispus (gm ODW/m^2)	Phosphorus in P. crispus (gm m^2)
30 March	0.53	50	0.27
20 April	0.38	70	0.27
25 May	0.35	120	0.42

During the early part of the period of rapid growth, P. crispus is utilizing its "luxury" levels of phosphorus absorbed during the winter and early spring (Table 18). Biomass accumulation continues into May after plant concentrations level off. At this time phosphorus must be absorbed from the environment. Two sources are available: the phosphorus-rich sediment and the water column.

The answer to where P. crispus gets its phosphorus is crucial to understanding the phosphorus budget of the lake. Laboratory studies using ^{32}P as a tracer have shown that young plants (20-30 cm) absorb $^{32}PO_4$ through their roots much more readily than they do through their leaves during short-term (24 hr) exposure (Birecka and Gebhardt 1980). During this period, very little phosphate was translocated from the absorbing organ. However, field data at West Station (Figure 32) shows large decreases in water total phosphorus levels in May, suggesting that P. crispus may be using phosphorus from the water column instead of that from the substrate.

Using biomass and plant phosphorus data for 1977 (the only year for which we have both data), we can estimate the magnitude of the decrease in phosphorus level in the water column that would be necessary if all of the phosphorus fixed into the P. crispus biomass between 20 April and 25 May came from the water. This decrease would be about 30 µg/l through the entire water volume of the lake. A decrease of this magnitude was seen in the springs of 1978 and 1979, though not in 1977, a year which saw much flooding and sediment input into the lake.

No matter what the source of phosphorus is for P. crispus during the short period of rapid growth in May, the sediment most likely supplies most of the phosphorus to the plant during its entire life cycle. P. crispus is obviously the source of the orthophosphate spikes in the shallow stations (Figure 14) in July. The net effect of this plant in Collins Lake must be a significant net transport of P from the sediment into the water column.

The growth burst of P. crispus in late April and early May occurs concomitantly with a decrease in total phosphorus in the deep water stations (Figure 32), a sharp increase in water clarity (Figure 23) and a decrease in chlorophyll a concentration (Figures 21, 22). The senscence, death, and decomposition of P. crispus in July is followed by increased phosphate concentration (Figures 14, 32), chlorophyll a concentrations and decreased water clarity. The timing of these events suggests a relationship.

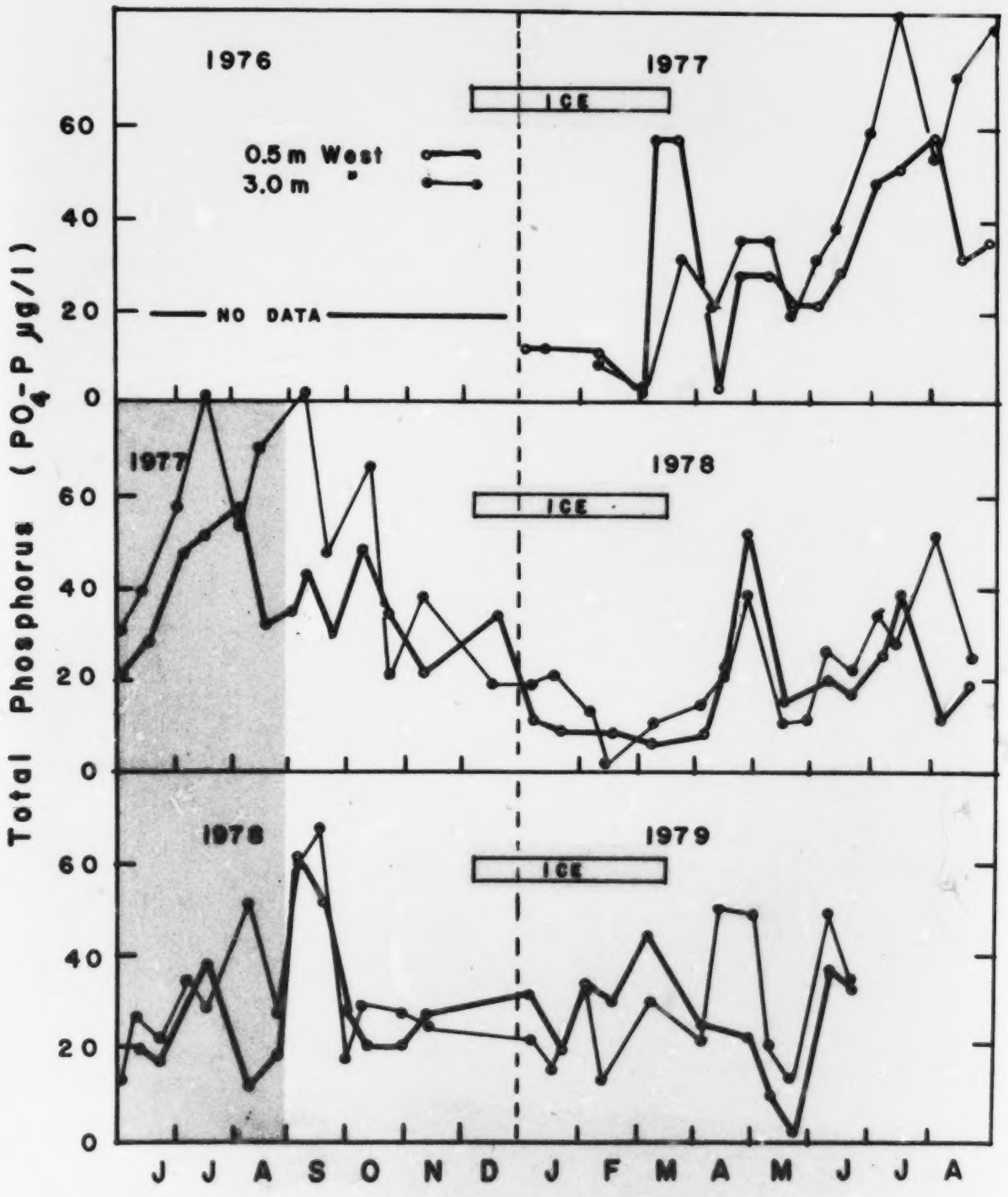


Figure 32. Total phosphorus (PO_4-P, µg/l) at 0.5 and 3.0 at West Station.

No experiments were carried out to determine which nutrients limit algal growth in Collins Lake, but indirect evidence implicates phosphorus. By comparing the sum of total Kjeldahl nitrogen (TKN) and NO_3-N with total phosphorus concentration of a given sample, the N:P ratio can be obtained. During the summer of 1978 and 1979 this ratio ranged in the epilimnion from 12 to 64 on a weight basis, with a mean of about 31. If the N:P ratio were calculated on an atomic basis, the mean would be about 68, which would put Collins Lake above the range of N:P ratios listed by Stumm and Baccini (1978) as typical for inland waters in general. Since the N:P atomic ratio in vigorously growing algal populations is typically 10 to 25 (Fuhs et al. 1972), the much higher ratio found in the water of Collins Lake indirectly suggests that phosphorus may be limiting the growth of the phytoplankton during the summer. This high N:P ratio may also explain the absence of the Cyanophyta during the summer.

The rapid increase in algal populations in early summer is most likely the result of the released nutrients from the decomposing P. crispus. The relationship of the rapidly growing P. crispus to the water phosphorus levels and algal population decline in the spring is still open to conjecture.

These data suggest that further dredging in Collins Lake would decrease the rate of sediment release of phosphorus not only by exposing sediments with low concentrations of available phosphorus, but also by reducing the role of P. crispus as a recycling agent. Reduced water phosphorus levels should decrease algal populations during the summer. The effect of the reduction of primary productivity by both the vascular plants and the phytoplankton on the lake ecosystem as a whole should be considered in the decision to dredge further.

REFERENCES

Allen, S. E. (ed) Chemical Analysis of Ecological Materials. John Wiley, N.Y. 565 pp.

Biggs, W. R. and J. H. Swinehart. 1976. Vanadium in selected biolgical systems. 141-196 pp. In: H. Sigel (ed) Metal ions in biological systems. Vol. 6 chapt. 2, Marcel Dekker, N.Y.

Birecka, H. and J. Gebhardt. 1980. The uptake and translocation of ^{32}P-labelled phosphate by the aquatic vascular plant, Potamogeton crispus L. Manuscript in review.

Bjork, Sven. 1977. Recovery and restoration of damaged lakes in Sweden, p. 110-133. In: J. Cairns, Jr., K. L. Dickson and E. E. Herricks (eds.) Recovery and restoration of damaged ecosystems. Univ. Pr. Virginia, Charlottesville.

Bjorkerud, S. 1963. The isolation of lipofuscin granules from bovine cardiac muscle, with observations on the properties of the isolated granules on the light and electron microscope levels. J. Ultrastructure Res. Suppl. 5, Acad. PG, New York, pp. 49.

Bogner, K. H., and A. E. Ellis. 1976. Properties and functions of lymphocytes and lymphod tissues in teleost fish. Fisch and Umvelt. Contrib. Histopath Fishes 4:59-72.

DiGiano, F. A. and P. D. Snow. 1977. Consideration of Phosphorus Release from Sediments in a Lake Model. In: Proceedings International Symposium-Interactions Between Sediments and Freshwater, UNESCO and Societas Internationalis Limnologiae, W. Junk, B. V. Publishers, The Hague, Netherlands.

Donato, H. and R. S. Sohal. 1978. Age-related changes in the lipofuscin-associated fluorescent substances in the adult male house fly, Musca domestica. Exp. Geront. 13(3-4):171-179.

Dunst, Russell C. (and 9 others). 1974. Survey of lake rehabilitation techniques and experiences. Dept. Nat. Res., Madison, Wisconsin, Tech. Bull. No. 75, 179 p.

Ellis, A. E., A. L. S. Munroe and R. J. Roberts. 1976. Defense mechanisms in fish. 1. A study of the phagocytic system and the fate of interperitoneally injected particulate material in the plaice (Pleuronectes platissa L.) J. Fish Biol. 8:67-78.

Fuhs, W., S. D. Demerele, E. Canelli and M. Chen. 1972. Characterization of phosphorus-limited plankton algae. p. 113-133. In: G. E. Likens (ed.) Nutrients and Eutrophication. Am. Soc. Limnol. and Oceanog., Special Symp. Vol. 1.

Gelin, C. 1978. The restoration of freshwater ecosystems in Sweden, p. 323-336. In: M. W. Holgate and M. J. Woodman. (eds.), The breakdown and restoration of ecosystem. Plenum Pr., New York.

Gyorgy, P. 1944. Experimental hepatic injury. American J. Clin. Path. 14: 67-88.

Gyorgy, P., and H. Goldblatt. 1941. Experimental production of dietary liver injury (necrosis, cirrhosis) in rats. Proc. Soc. Exper. Biol. med. 46:492.

Hartroft, W. S., and E. A. Porta. 1965. Ceroid. American J. Med. Sci. Path. Bact. 250:3245.

Heidinger, R. C., and S. D. Crawford. 1977. Effects of temperature and feeding rate on the liver-somatic index of the largemouth bass, Micropterus salmonides. J. Fish. Res. Board. Canada 34:

Larson, G. L. 1973. Liver weight of brook trout in a high-mountain lake in Washington state. Progr. Fish Cult. 35(4):234-236.

Lillie, R. D., F. S. Daft, and W. H. Sebrell, Jr. 1941. Cirrhosis of the liver in rats on a deficient diet and the effect of alcohol. Publ. Health Rep. 56:1255-1258.

Munnell, J. F., and R. Getty. 1968. Rate of accumulation lipofuscin in the canine heart. J. Geront. 23:154.

Pearse, A. G. Everson. 1968 (3rd ed.) Histochemistry, Theoretical and applied, Vol. 1. Churchill Livingston, Edinburgh, London and New York. 759 p.

Peterson, Spencer A. 1979. Limnological and socioeconomic evaluation of lake restoration projects: approaches and preliminary results. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon EPA-600/3-79005.

Prescott, G. W. 1962. Algae of the western Great Lakes area. W. C. Brown, Dubuque, Iowa. 977 p.

Reichel, W. 1968. Lipofuscin pigment accumulation and distribution in five rat organs as a function of age. J. Geront. 23:145.

Reichel, W., J. Hollander, J. H. Clark and B. L. Strehler. 1968. Lipofuscin pigment accumulation as a function of age and distribution in rodent brain. J. Geront. 23:71-78.

Roberts, R. J. 1975. Melanin-containing cells of teleost fish and their relationship to disease, pp. 399-428. Ribelin and Migaki (eds.) The pathology of fishes. Univ. Wisc. Pr., Wisc.

Snow, P. D. and F. A. DiGiano. 1976. Mathematical Modeling of Phosphorus Exchange between Sediments and overlying water in shallow Eutrophic Lakes, Report No. Env. E. 54-76-3. Mass. Water Pollution Control, Mass. Water Resources Comm., pp. 225 (based on Ph.D. Dissertation, Univ. of Mass.)

Sorenson, E. M. B. 1976. Ultrastructure changes in the hypotocytes of green sunfish, Lepomis cyanellus Rafinesque, exposed to solutions of sodium arsenate. J. Fish. Biol. 8:229-240.

Standard Methods for the Examination of Water and Wastewater, 1975, 14th edition, 1193 pp.

Stumm, W., and P. Baccini. 1978. Man-made chemical desturbations of lakes. p. 91-126. In: A. Lerman (ed.) Lakes: chemistry, ecology. physics. Springer Verlag, N. Y. 363 pp.

Theis, Thomas L., and Joseph V. DePinto. 1976. Studies on the reclamation of Stone Lake, Michigan. U.S. Environmental Protection Agency, EPA-600/3-76-106, 84 p.

Valtonen, T. 1974. Seasonal and sex-bound variation in the carbohydrate metabolism of the liver of the whitefish. Comp. Biochem. Physio. 47(A): 713-722.

Vollenweider, R. A. 1974. Primary Production in Aquatic Environments. IBP Handbook No. 12. Blackwell Scientific Publications. 225 pp.

Wehrmeister, J. R. 1978. An ecological life history of the pondweed Potamogeton crispus L. in North America. Ohio State University Center for Lake Erie. Area Research, Columbus, Ohio. Clear Technical Report 49. 157 pp.

Welch, C., C. George, P. D. Snow, P. Tobiessen and J. Bloomfield. 1979. The resotration of Collins Lake, Scotia, New York. Part I-design, implementation and preliminary evaluation. Clean Lakes Program, U.S. Environmental Protection Agency, Washington, D.C. 20469. 60 p.

Wetzel. R. A. 1975. Limnology. W. B. Saunders, Co. Phila. 743 pp.

Woodard, S. A. 1978. The relationship between the macroinvertebrates and the senescence, death and decay of the aquatic macrophyte. Potamogeton crispus in Collins Lake. Union College MS Thesis. 79 pp.

Wood, E. M., and W. T. Yasutake. 1956. Ceroid in fish. American J. Path. 32(3):591-603.

APPENDIX A. Students involved in gathering and interpreting data for this report.

Physical and Chemical Analysis: Timothy Pangburn, Steven Gyory, Andrew Adriance, George Wells, Timothy Sullivan, Mark Massey, Rodney Aldrich, Douglas Howie, Daniel Breg, Mark Feinberg, Paul VanLaak, Robert Schick, David Cornell, Philip Hood, Timothy Baldwin, Paul Spence, June Rinkoff, David Jones, Esther Quirk.

Computer Analysis: David Jones, Thomas Hallenbeck, James Lerner, David Powell, Matthew Metcalf, Barry C. Miller.

Phyloplankton Studies: Marion Baciewicz, Joyce Warner, Timothy Cross, Gloria Ornvold, William Colburn, Lily Pak.

Zooplankton Studies: Katherine Hollister, Adam Berg, Robert Lein, Paul McLaughlin, Bruce Levin, Joanne Haddad, Janice Fischer.

Vascular Plant Studies: Paul Powers, Edward Hayes, Gary Geller, Barbara Carey, Christine Vilardi, Bonnie Sendzicki, Joan Gebhardt, Alan Woodard, Richard Hall.

Ichthyology: Carol Grant, Denise Polsinelli, Thomas D. Engel, Laurence M. Merlis, Glenn R. Delaney, Carmen A. Gatta, Iaian A. Drummond, Christopher L. Brown, Mark A. Klemperer, Dennis D. Curtin, John M. Foehl Jr., Francesca Messina, John H. Samuelian, Lynn Laffin, Eric J. Goldring, John Llyons, David Philippe Beguin, Steven A. Lacy, Terri M. Moran, Debra Kaufman, Ron Shelton, Mark M. Kawczynski.

APPENDIX B. Species list of all phytoplankton found in Collins Lake. Nomenclature follows Prescott (1962). Capitalized abbreviations refer to Appendix C, Plankton Data Summary

	Abbr.
DIVISION CHLOROPHYTA	
Ankistrodesmus falcatus (Corda) Kalts, 1848	ANK
Carteria globulosa, Pascher	
Carteria Klebsii (Dang.) Dill, 1895	
Cerasterias stauroides, West & West, 1895,	
Chlamydomonas epiphytica, G.M. Smith, 1920	CME
globosa, Snow, 1903	
Snowii, Printz, 1914	
Chlamydomonas spp.	CHL
Chlorella vulgaris, Beyerinck, 1890	
Closteriopsis longissima, Lemmerman, 1899a	
Closterium spp.	
Coelastrum reticulatum (Dang.) Senn, 1899	RET
Cosmarium spp.	
Crucigenia apiculata (Lemm.) Schmidle, 1901	CRA
rectangularis (A. Braun) Gay, 1891	CRR
quadrata, Morren, 1830	CRQ
Dictyosphaerium pulchellum, Wood, 1874	DIC
Elaktothrix gelatinosa, Wille, 1898	
viridis (Snow) Printz, 1914	ELV
Franceia Droescheri (lemm.) G.M. Smith, 1933	
Gloeocystis planctonica (West & West) Lemmerman, 1915	GLP
Kirchneriella subsolitaria, G.S. West, 1908	
Kirchneriella spp. Schmidle, 1893	KIR
Nephrocytium spp. Naegeli, 1849	
Oocystis parva, West & West, 1898	OOP
pusilla, Hansgirg, 1890	
solitaria, Wittrock & Nordstedt, 1879	OOS
Oocystis spp.	
Pediastrum simplex (Meyen) Lemmerman, 1897	PED
tetras (Erenb.) Ralfs	
Phacotus spp.	PHA
Pyramimonas spp., Schmarda, 1850	
Schroederia judayi, G.M. Smith, 1916	SCH
Scenedesmus bijuga var. alternas (Reinsch) Hansgirg, 1888	SCB
dimorphus (Turp.) de Brébission, 1835	
quadricanda (Turp.) de Brébission, 1835	SCQ

DIVISION CHLOROPHYTA (continued)

Selenastrum minutum (Naeg.) Collins, 1909	SEL
Sphaerocystis Schroteri Chodat, 1897	SPH
Staurastrum chaetoceras (Schröder) G.M. Smith	
Staurastrum spp.	
Teträedron caudatum (Corda.) Hansgirg, 1888a	TEC
minimum (A. Braum) Hangirg, 1888a	
trigonum var. gracile (Reinsch) DeToni, 1889	
Volvox spp. Linnaeus, 1758	

DIVISION CHRYSOPHYTA

Achnanthes spp.	
Amphipleura spp.	
Amphora ovalis	
Asterionella formosa Hass.	AST
Chaetoceros spp.	
Chromulina spp.	
Cocconeis spp.	
Cyclotella antiqua	
Cyclotella spp.	CYC
Diatoma spp.	
Dinobryon bavaricum, Imhof, 1890	DIB
sertularia, Ehrenberg, 1835	DIN
Fragilaria spp.	
Frustulia spp.	
Gomphonema spp.	
Gyrosigma spp.	
Mallomonas app. Perty, 1852	MAL
Melosira granulata	MEL
Melosira, spp.	
Navicula, spp.	NAV
Nitzschia, spp.	
Surirella spp.	
Stephanodiscus spp.	
Synedra spp.	SYN
Tabellaria spp.	

DIVISION CYANOPHYTA

Agmenellum quadruplicatum (Meneghini) Brébission	
Anabaena spinoides	
Anabaena sp.	ANA
Chroococcus dispersa (Keissl.) Lemmerman, 1904	COD
Chroococcus spp.	COO
Coelosphaerium Naegelianum Unger	
Gloecapsa punctata Naegeli, 1849	
Merismopedia tenuissima Lemmerman, 1898d	MER
Microcystis aeruginosa Kuetz, Elenkin, 1924	

DIVISION CYANOPHYTA (continued)

Microcystis spp. Kuetzing, 1833a	MIC
Oscillatoria articulata Gardner	
rubescens DeCandolle, 1825	
Oscillatoria spp. Vaucher, 1803	OSC
Spirulina spp. Turpin, 1827	
Ceratium hirundinella (D.F. Muell) Dujardin, 1841	CER
Crytomonas erosa Ehrenberg, 1838	CME
ovata Ehrenberg, 1838	CMO
pusilla	CMP
Glenodinium spp. (Ehrenb.) Stein, 1883	GLN
Gymnodinium spp. (Stein) Kofoid & Swezy, 1921	

DIVISION EUGLENOPHYTA

Euglena spp.	
Phacus longicauda (Ehrenb.) Dujardin, 1841	
Phacus spp.	
Trachelomonas varians (Lemm.) Deflaudre, 1924	
Trachelomonas spp.	TRA

APPENDIX C. Plankton Data Summary for South Station 0.5 m. South Station was moved into the dredged area in November 1977. Abbreviations for the species can be found in Appendices B and D. Numbers in parentheses are densities in cell/l in exponential notation (3e7 = 3 x 10^7) or animals/l. Within each group, species are ranked with #1 having the greatest density. Phyto = phytoplankton, Zoo = zooplankton.

Date		6/22/76	6/29/76	7/6/76	7/13/76	7/20/76	7/27/76	8/3/76
Phyto	1	CMO(3e6)	PED(1e7)	DIB(2e7)	PED(1e7)	No Data	CRQ(2e7)	PED(1e7)
	2	PED(4e5)	OOC(3e6)	PED(6e6)	CRA(4e6)		PED(2e7)	DIC(5e6)
	3	CMP(1e5)	COE(1e6)	SPH(3e6)	OOC(3e6)		DIC(1e7)	OOC(3e6)
	4	SCE(9e4)	SCE(6e5)	OOC(1e6)	CRQ(2e6)		OOC(1e7)	SCE(3e6)
Zoo								
Cope	1	-	CBT(22.2)	CBT(33.3)	MED(77.7)	DB (22.2)	MED(11.1)	-
	2	-	-	-	-	MED(22.2)	-	-
Clad	1	SK (2.8)	CRR(33.3)	EUC(66.6)	EUC(577)	EUC(233)	-	-
	2	ALC(2.8)	EUC(11.1)	CRR(111)	CRR(55.5)	CHS(156)	-	-
Rot	1	KC (27.7)	MOQ(11.1)	KC (378)	KC (489)	KC (156)	KC (111)	KC (300)
	2	COU(13.8)	-	COU(211)	ASP(122)	ASP(156)	BRU(178)	ASP(167)
	3	PLY(8.3)	-	PLY(66.6)	COU(88.8)	BRU(55.5)	ASP(111)	FL(55.5)

Date		8/11/76	8/19/76	8/31/76	9/14/76	10/9/76	10/28/76	11/11/76
Phyto	1	No Data	PED(2e7)	PED(2e7)	PED(5e6)	AST(6e6)	AST(6e6)	AST(2e6)
	2		MEL(6e6)	DIC(2e6)	CRQ(5e5)	AST(7e5)	PED(6e5)	PED(1e5)
	3		DIC(4e6)	SCE(1e6)	SCE(3e5)	CMP(8e4)	CMP(3e5)	MEL(5e4)
	4		SCE(2e6)	MIC(1e6)	COE(2e5)	MEL(7e4)	ANK(6e4)	ANK(1e4)
Zoo								
Cope	1	-	DB (11.1)	MED(22.2)	-	DB (5.6)	DB (2.3)	DB (2.3)
	2	-	-	DB (11.1)	-	-	-	-
Clad	1	-	-	DBM(77.7)	-	EUC(19.4)	EUC(2.3)	DGM(4.5)
	2	-	-	-	-	-	-	EUC(9.1)
Rot	1	KC (244)	KC (967)	KC (133)	-	KC (50)	KC (22.7)	KC (50)
	2	KL (33.3)	ASP(533)	KLL(22.2)	-	FL (5.6)	PLY(4.5)	KLL(2.3)
	3	-	KLL(200)	FL (11.1)	-	KLL(2.8)	-	KLB(2.3)

APPENDIX C (continued)

Date		1/5/77	2/2/77	2/16/77	3/2/77	3/30/77	4/13/77	4/27/77
Phyto	1	CMP(2e5)	PED(6e4)	No Data	PED(6e4)	CHG(5e5)	AST(3e5)	SPH(2e5)
	2	NAV(4e4)	-		TRA(6e4)	SCH(2e5)	PED(1e5)	CRQ(2e5)
	3	PED(4e4)	-		CHL(2e4)	CHL(1e5)	CHL(1e5)	PED(9e4)
	4	CMO(2e4)	-		SEL(2e4)	PED(1e5)	SCH(1e5)	SYN(8e4)
Zoo								
Cope	1	-	CBT(2.8)	CBT(218)	-	BD (2.8)	DB (2.3)	-
	2	-	-	-	-	-	CBT(4.5)	-
Clad	1	DGM(2.3)	-	DGM(8.3)	-	-	-	CHS(22.2)
	2	-	-	-	-	-	-	-
Rot	1	NSQ(36.4)	KC (11.1)	KC (11.1)	KC (6.8)	KC (22.2)	MOL(11.4)	KC (392)
	2	NAC(4.5)	KLL(2.8)	KLL(2.8)	-	-	NSQ(2.3)	CPH(256)
	3	KC (36.4)	PLY(2.8)	-	-	-	KC (50)	KLL(80.6)

Date		5/11/77	5/25/77	6/8/77	6/22/77	7/6/77	7/20/77	8/3/77
Phyto	1	FRA(1e6)	No Data	No Data	No Data	PED(4e6)	PED(2e7)	PED(4e6)
	2	SYN(5e5)				SPH(2e6)	COE(7e5)	SCE(4e5)
	3	TAB(4e5)				OOC(2e6)	KIR(3e5)	KIR(2e5)
	4	SCE(1e5)				CMP(6e5)	MEL(2e5)	CRQ(1e5)
Zoo								
Cope	1	-	MED(5.6)	-	MED(11.1)	MED(2.8)	DB	-
	2	-	-	-	CBT(8.3)	-	-	-
Clad	1	CHS(22.2)	CHS(41.7)	EUC(4.5)	CHS(5.6)	DGM(8.3)	-	-
	2	-	-	CHS(2.3)	CRD(11.1)	CRP(8.3)	-	-
Rot	1	KC (214)	KC (253)	KC(118)	KC(41.7)	CPH(2.8)	KC (50)	KLL(211)
	2	MOL(44.4)	PLY(181)	PLY(40.9)	PLY(11.1)	KC (8.3)	PLY(2.3)	KC(125)
	3	LOV(13.9)	MOC(69.4)	MOL(15.9)	ASP(5.6)	-	-	FL (54.5)

APPENDIX C (continued)

Date		8/17/77	8/31/77	9/14/77	9/28/77	10/12/77	10/26/77	11/9/77
Phyto	1	PED(1e6)	PED(6e6)	PED(6e6)	PED(2e6)	PED(7e5)	OSC(8e5)	CMP(5e5)
	2	ANA(4e5)	SCE(6e5)	OSC(4e5)	OSC(6e5)	MAL(2e5)	CMP(1e5)	DIB(3e5)
	3	SCE(3e5)	KIR(4e5)	OOC(3e5)	CRQ(2e5)	CMP(1e5)	DIN(9e4)	MAL(6e4)
	4	DIN(2e5)	MEL(1e5)	SCE(1e5)	SCE(1e5)	MEL(6e4)	PED(8e4)	ANK(4e4)
Zoo								
Cope	1	MED(2.3)	-	-	-	-	DB (2.3)	DB (22.7)
	2	DB (2.3)	-	-	-	-	-	CBT(6.8)
Clad	1	DP (2.3)	-	EUC(2.3)	EUL(2.3)	CHS(2.3)	CHS(2.3)	EUL(6.8)
	2	-	-	-	DGM(2.3)	-	DP (2.3)	-
Rot	1	KC(63.6)	KC(493)	PLY(2.3)	PLY(18.2)	KC (25)	KC(38.6)	KC(293)
	2	FL(43.2)	PLY(38.6)	KC(45.5)	KC(47.7)	PLY(6.8)	PLY(40.9)	KLL(11.4)
	3	KLL(9.1)	FL(22.7)	-	-	-	CPH(4.5)	PLY(11.4)

Date		12/28/77	1/11/78	2/10/78	2/22/78	4/5/78	4/19/78	5/3/78
Phyto	1	COO(7e4)	CMP(2e4)	CMP(4e4)	NAV(2e3)	COO(7e5)		CMP(5e5)
	2	CME(2e4)	PED(2e4)	CMO(2e4)	-	PED(2e5)	No Data	TRA(4e5)
	3	PED(8e3)	NAV(1e3)	AST(5e3)	-	FRA(8e4)		FRA(1e5)
	4	AST(5e3)	-	TRA(4e3)	-	TRA(4e4)		PED(1e5)
Zoo								
Cope	1	DB(4.5)	CBT(4.5)	DB(4.5)	CBT(2.3)	CBT(2.3)	-	-
	2	-	-	-	-	-	-	-
Clad	1	-	-	-	-	-	EUC(2.3)	-
	2	-	-	-	-	-	CHS(2.3)	-
Rot	1	KLL(31.8)	KLL(4.5)	KLL(15.9)	KLL(2.3)	PLY(2.3)	KC (2.3)	PLY(2.3)
	2	KC(18.2)	-	KC(2.3)	NAC(2.3)	KC (2.3)	KC (6.8)	NSQ(2.3)
	3	KLB(2.3)	-	KLB(2.3)	-	KLL(2.3)	-	KLL(2.3)

APPENDIX C (continued)

Date		5/17/78	5/31/78	6/13/78	6/28/78	7/12/78	7/18/78	7/26/78
Phyto	1	OSC(1e6)	CMP(3e6)	CMP(136)	OOC(4e6)	PED(1e7)		PED(836)
	2	TRA(2e5)	PED(2e5)	PED(1e6)	DIC(1e6)	OOP(1e7)	No Data	CMP(1e6)
	3	SCE(1e5)	OOC(2e5)	SPH(3e5)	CMP(1e6)	TRA(6e6)		SCE(1e6)
	4	FRA(1e5)	OSC(1e5)	COO(1e5)	PED(1e6)	OOC(2e6)		COO(9e5)
Zoo								
Cope	1	-	MED(9.1)	MED(2.3)	DB (34.1)	DB (2.3)	-	DB (2.3)
	2	-	CBT(2.3)	DB (2.3)	MED(4.5)	-	-	-
Clad	1	CHS(2.3)	CHS(13.6)	EUL(20.5)	CRD(511)	EUL(13.6)	-	-
	2	-	-	CRP(4.5)	EUL(105)	CRD(6.8)	-	-
Rot	1	CPH(2.3)	LPA(9.1)	COH(34.1)	COH(632)	CK(1110)	KC(307)	FL(773)
	2	NSQ(2.3)	KC (11.4)	KC (22.7)	PLY(398)	PLY(335)	PLY(6.8)	KC(1440)
	3	KC (2.3)	CPH(4.5)	MOC(4.5)	KC (193)	FL (50)	FL (40.9)	PLY(29.5)

Date		8/9/78	8/23/78	9/5/78	9/20/78	10/4/78	10/18/78	11/1/78
Phyto	1	PED(1e7)	PED(2e7)	PED(3e7)	PED(1e7)	PED(8e6)	CMP(4e6)	DIC(2e6)
	2	SCE(2e6)	CMP(1e6)	COD(3e6)	MER(2e6)	CMP(1e6)	PED(2e6)	PED(8e5)
	3	CMP(1e6)	SPH(9e5)	CMP(1e6)	GLP(1e6)	DIC(7e5)	DIC(9e5)	CMP(8e5)
	4	SPH(9e5)	SCE(9e5)	CRQ(7e5)	SCE(6e5)	SCE(4e5)	COO(8e5)	TRA(2e5)
Zoo								
Cope	1	DB (6.8)	MED(9.1)	DB (11.4)	DB (118)	DB (38.6)	DB (30)	DB (6.8)
	2	-	DB (15.9)	-	MED(9.1)	MED(2.3)	-	-
Clad	1	EUC(54.5)	DL(6.8)	-	DL (27.3)	EUC(31.8)	EUC(9.1)	EUC(29.5)
	2	CRD(6.8)	DGM(4.5)	-	EUC(20.5)	DGM(25)	-	-
Rot	1	KC(1520)	KC(1660)	KC(1940)	KC (209)	KC (261)	KC (752)	KC (252)
	2	FL (107)	FL (109)	KLL(1980)	KLL(63.6)	PLY(56.8)	PLY(177)	KLL(27.3)
	3	KLL(102)	KLL(259)	PLY(63.6)	ASP(15.9)	KLL(6.8)	KLL(13.6)	PLY(61.4)

APPENDIX C (continued)

Date		11/15/78	1/10/79	1/24/79	2/7/79	2/21/79	3/7/79	4/4/79
Phyto	1	COO(1e6)	CME(4e5)	COO(1e6)	CMP(7e5)	DAC(3e6)	CMP(6e5)	CMO(5e6)
	2	PED(9e5)	CHL(3e5)	CMP(2e5)	ANA(6e5)	CMP(9e5)	TRA(5e5)	CMP(5e6)
	3	CHL(3e5)	GLP(1e5)	CME(4e4)	COO(4e5)	COO(6e5)	CHL(5e5)	TRA(1e6)
	4	TRA(3e5)	CMP(1e5)	TRA(2e4)	CMO(2e5)	CHL(3e5)	DAC(3e5)	AST(1e6)
Zoo								
Cope	1	CBT(22.7)	CBT(63.6)	CBT(52.3)	CBT(25)	CBT(13.6)	-	CBT(61.4)
	2	DB (11.4)	DB (2.3)	-	-	-	-	-
Clad	1	EUC(40.9)	EUC(4.5)	EUC(2.3)	-	EUC(2.3)	CHS(2.3)	EUL(4.5)
	2	DGM(6.8)	DGM(2.3)	-	-	CHS(2.3)	-	EUC(4.5)
Rot	1	KC (173)	KC (123)	KC (22.7)	KC (27.3)	KC (6.8)	-	KC (40.9)
	2	KLL(68.2)	KLL(18.2)	KLL(6.8)	KQ (2.3)	COL(4.5)	-	KQ (18.2)
	3	KLB(34.1)	NSQ(6.8)	NSQ(4.5)	KLL(2.3)	NSQ(2.3)	-	NSQ(13.6)

Date		4/18/79	5/2/79	5/16/79	5/30/79	6/13/79	*6/27/79	*7/11/79
Phyto	1	CMP(2e6)	COO(3e6)	COD(9e5)	CLP(3e6)	CMP(9e5)	PED(4e6)	PED(6e6)
	2	TRA(7e5)	CHL(6e5)	PED(5e5)	SPH(2e6)	PED(7e5)	CMP(2e6)	CMP(1e6)
	3	CME(7e5)	CME(3e5)	CLP(4e5)	CMP(7e5)	CHL(3e5)	OOS(1e6)	RET(1e6)
	4	AST(3e5)	SYN(2e5)	KIR(4e5)	CHL(4e5)	SCH(1e5)	OOC(835)	OOC(1e6)
Zoo								
Cope	1	CBT(9.1)	DB (13.6)	CBT(15.9)	DB(38.6)	MED(38.6)	MED(11.4)	MED(6.8)
	2	-	CBT(11.4)	DB (6.8)	CBT(9.1)	DB (20.5)	DB (4.5)	CBT(2.3)
Clad	1	CHS(4.5)	CHS(6.8)	EUL(13.6)	DGM(43.2)	CRD(93.2)	DL(20.5)	DGM(15.9)
	2	EUC(2.3)	EUL(2.3)	DGM(4.5)	ALC(6.8)	EUL(25)	DGM(18.2)	CRC(15.9)
Rot	1	KC (30)	KC (716)	KC (789)	KC (339)	KC (233)	KC (143)	KC (870)
	2	KQ (11.4)	KQ (16.4)	KQ (221)	KQ (293)	COH(100)	PLY(93.2)	PLY(182)
	3	NSQ(4.5)	PLY(9.1)	KLB(2.3)	LPA(15.9)	PLY(27.3)	COH(65.9)	MOC(2.3)

APPENDIX C (continued)

Date		7/25/79	8/8/79	8/22/79
Phyto	1	PED(1e7)	PED(1e7)	PED(7e6)
	2	COD(1e6)	CRA(7e5)	TRA(4e5)
	3	OOP(1e6)	TRA(5e5)	KIR(4e5)
	4	GLP (6e5)	SCE (5e5)	FRA (3e5)
Zoo				
Cope	1	-	MED(9.1)	DB (15.9)
	2	-	-	MED(18.2)
Clad	1	DGM(9.1)	DGM(22.7)	DL (11.4)
	2	CRD (2.3)	DL(9.1)	DMG(6.8)
Rot	1	KC (4090)	KC (8320)	KC (155)
	2	PLY(86.4)	FL(564)	PLY(150
	3	FL(11.4)	PLY(500)	FL(2.3)

APPENDIX D. ZOOPLANKTON SPECIES LIST. ABBREVIATIONS REFER TO SPECIES FOUND IN APPENDIX C.

Cladocera	
Leptadoridae	
Leptadora kindtii (Focke)	(LK)
Sididae	
Sida crystallina (Müller)	(SIC)
Diaphanosoma leuchtenbergianum (Fischer)	(DL)
Daphnidae	
Daphnia galeata mendotae (Sars, Birget)	(DGM)
parula (Fordyce)	(DP)
Ceriodaphnia reticulata (Jurine)	(CRR)
pulchella (Sars)	(CRP)
Simiocephalus cerrulatus (Koch)	(SS)
vetulus (Schodler)	(SV)
Scapholebris kingi (Sars)	(SK)
Bosminidae	
Eubosmina coregoni (Baird)	(EUC)
longispina (Leydig)	(EUL)
Eurycercinae	
Eurycercus lamellatus (Muller)	(ERL)
Chydorinae	
Camptocerous rectirostris (Schodler)	(CRC)
Alona costata (Sars)	(ALC)
guttata (Sars)	(ALG)
Chydorus globosus (Barid)	(CHG)
sphaericus (Müller)	(CHG)
Plexorus denticulatus (Birge)	(PLD)
Copepoda	
Diaptomidae	
Diaptomus birgei (Marsh)	(DB)
Cyclopidae	
Eucyclops agilis (Koch)	(EUA)
Cyclops vernalis (Fischer)	(CYV)
bicuspidatus thomasi (Forbes)	(CBT)
Mesocyclops edax (Forbes)	(MED)
Rotifera	
Brachionidae	
Brachionus angularis (Gosse)	(BRA)
patulus (Wulfert)	(BRP)
urceolaris (Müller)	(BRU)
quadridentata	(BRQ)

APPENDIX D (continued)

Kellicottia longispina (Kellicott)	(KLL)
bostoniensis (Rousselet)	(KLB)
Keratella cochlearis (Gosse)	(KC)
quadrata (Müller)	(KQ)
Nothalca squamula (Müller)	(NSQ)
acuminata (Ehrenberg)	(NAC)
Colurella sp. (Bory de St. Vincent)	(COL)
Lepadella patella (Muller)	(LPA)
ovalis (Muller)	(LOV)
Lecane luna (Muller)	(LL)
ohioensis	(LO)
depressa	(LD)
Monostyla quadridentata	(MOQ)
lunaris	(MOL)
closterocera	(MOC)
Notommatidae	
Cephalodella sp.	(CPH)
Monommata sp.	(MON)
Trichoceridae	
Trichocera sp.	(TRC)
Asplanchnidae	
Asplanchna priodonta (Gosse)	(ASP)
Synchaetidae	
Polyarthra vulgaris (Carlin)	(PLY)
remata (Skorikow)	
euryptera (Wierzejski)	(PLE)
Testudinellidae	
Testudinella mucronata	(TEM)
patina	(TEP)
intermedia	(TEI)
Filinia longiseta (Ehrenberg)	(FL)
Cochilidae	
Conchilus hippocrepis (Schrank)	(COH)
unicornis (Rousselet)	(COU)